ESTIMATION OF OPTIMUM COMPACTION LEVEL FOR BIOREACTOR LANDFILL OPERATION

by

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Abstract

ESTIMATION OF OPTIMUM COMPACTION LEVEL FOR BIOREACTOR LANDFILL OPERATION

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The effect of compaction on the degradation of MSW is important for the successful operation of bioreactor landfill. It has been a general tendency to compact working face solid waste as much as possible to gain more space. Previous studies has shown moisture content, moisture recirculation and ambient temperature were important components for degradation. There was a limitation on the compaction effect on degradation. Although several studies were conducted in the past to find the hydraulic conductivity of the waste, there was lack of work to find the optimum compaction to achieve sufficient flow inside the landfill and thus to enhance the biodegradation process.

The current study has been focused on the compaction effect on the hydraulic properties of various waste and determining optimum range of density to maintain the flow in fresh waste. A certain compaction level is required for bioreactor landfill which can maintain uniform flow with sufficient drainage capacity. Besides flow criteria, compaction level is required for the microbial activity and can maximize methane gas production. Drainage capacity of the solid waste is an important parameter for the bioreactor landfill which is required to generate sufficient leachate. The tangent intersection methods were used to estimate the changing point on the curves, which were considered as optimum density. They hydraulic conductivity versus dry density gave the optimum density range from 482 to 520 kg/m^3 and the corresponding range of hydraulic conductivity varied from

7.0×10^{-4} to 1.1×10^{-3} cm/s. The retained porosity versus dry density gave the optimum density range from 490 to 520 kg/m^3 for various waste. The maximum retained porosity 46% to 51% for various waste. Similarly, the drainable porosity versus dry density gave the optimum density from 480 to 520 kg/m^3 for various waste. The minimum drainable porosity was estimated as from 9% to 12% for various waste using different devices.

Similarly, the study had been focused on the compaction effect on the degradation. MSW samples with the same physical characteristics were filled in three bioreactor cells at dry densities of 458, 572 and 686 kg/m^3 designated as R1, R2 and R3, respectively. The rate of methane gas generations for compacted samples for reactor R1, R2 and R3 reached peak 315, 245 and 77 mL/kg/day, respectively. There was not big difference on methane gas generation between reactors R1 and R2 but reactor R3 produced very low methane. The reactors R1, R2 and R3 produced cumulative methane 52, 39 and 12.5 L/kg, respectively. The compaction ratio of reactors R1, R2 and R3 were in the ratio of 1:1.25:1.5, and the gas generation ratio as 4.16:3.12:1, respectively. The degradation level in reactors R1 and R2 were quite high as compared to reactor R3. Even though there was high degradation in reactors R1 and R2, the hydraulic conductivities were not varying but almost same in every month. There was not much degradation on the highly compacted waste reactor R3 but the hydraulic conductivity decreased every month which indicated that hydraulic conductivity decreased due to accumulation of gas rather than degradation by itself. It was general assumption that if the leachate generation is high, gas generation will also be high. The leachate generation was also decreased with compaction. While considering all above criteria to estimate the optimum density level, the dry density should be less than that of reactor R2 i.e. the dry density of 572 kg/m^3.

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Chapter 1

Introduction

1.1 Background

The Solid Waste Association of North America (SWANA) defines bioreactor landfill 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. The bioreactor landfill significantly increases the extent of organic waste decomposition, conversion rates, and process effectiveness over what would otherwise occur with the landfill.” Water is the key factor to accelerate the biochemical decomposition of organic substances (Pohland 1970; Lechie et al., 1979; Klink and Ham 1982).

The concept of operating the landfill as a bioreactor emerged from the addition of moisture that stimulates microbial activity by providing better contact between the waste and microorganism via solvent medium. Numerous advantages are attributed to the operation of a landfill as a bioreactor including accelerated biological decomposition that have been reported in literature as: (1) decomposition and biological stabilization occurs at an accelerated rate achieving majority of the settlement before placement of the final cover which decreases the risk of damage to the final cover; (2) increased effective MSW density and space gain due to enhanced degradation rate; (3) in-situ leachate treatment and the reduction of leachate handling cost; (4) increased rates of gas production; (5) accelerated MSW decomposition process reduce post-closure care time frames, monitoring requirements and costs of the landfill (Barlaz et al. 1990; Reinhart and Townsend 1997). Bioreactor landfills are designed and operated to optimize the waste stabilization process rather than to simply contain the wastes, as is prescribed by most regulations (Reinhart and Townsend, 1997; Reinhart et al., 2002). The most common strategy to accelerate decomposition is to stimulate microbial activity by adding moisture to the waste via recirculation of leachate and addition of supplemental liquids, a practice that is becoming more common in North America (Pohland 1975; Barlaz et al. 1990; Pacey et al. 1999; Reinhart et al. 2002; Mehta et al. 2002; Warith 2002; Benson et al. 2007; Bareither et al. 2008a). Higher rates of MSW biodegradation eventually cause a reduction of the contaminant life span of the landfill and thus decrease in the cost of long term monitoring (Warith 2002).

Circulation of water/leachate into waste is most important task in a bioreactor landfill so that understanding of the permeability become necessary. Basically hydraulic conductivity of any materials is dependent on its porosity. There have been a number of studies on the hydraulic conductivity of the MSW performed both in the laboratory and in the field scale and the reported values of hydraulic conductivities ranges from 10^{-2} to 10^{-7} cm/sec. Although several studies were conducted in the past to find the hydraulic conductivity of the waste, there was lack of systematic work to find the optimum compaction which can maintain flow and thus to enhance the microbial activity. Liquid flow inside the waste takes place both in saturated and unsaturated condition. The study is focused on the effects of density, composition and degradation on the hydraulic conductivity of the waste. Several researchers (Tiquia et al., 1998; McKinley et al., 1986; Suler and Finstein, 1977) have already recommended water is required to enhance the biodegradation and proposed 40-60% to be the optimum moisture content level for the waste. Optimum moisture content might be one of the key factors for facilitating degradation of the waste. The compaction for the municipal solid waste should be determined which can maximize methane gas generation without minimizing the required flow while doing recirculation of leachate.

1.2 Problem statement

The waste generated from the different residential areas goes to the landfill. Landfill operators have general tendency to compact working face solid waste as much as possible to gain more space. It is obvious that both over and under compaction are not desirable for the successful bioreactor landfill operation. Higher density maximizes the available space while, loosely compacted waste occupies large volume. However, too dense and too loose compactions both are not desirable for bioreactor landfill operation. The higher compaction may lead to decrease in total and drainable porosity, hydraulic conductivity of solid waste, and eventually hinders the liquid flow pattern within the landfill. During bioreactor landfill operation, additional water/leachate is added to increase microbial activities and waste decomposition. Higher compaction may lead to failure in bioreactor landfill operation due to (1) insufficient moisture flow within the landfilled solid waste, (2) creation of preferential liquid flow channel and flow towards the slope, (3) decrease in gas production and gas flow within the solid waste (4) create problem in leachate generation and collection. On the other hand, loosely compacted solid waste may create stability problem while pumping the water/leachate. Generally loosely compacted waste has a high field capacity and thus creates high unit weight with low shear strength within the waste mass. Koerner and Soong (2000) reported ranges of landfill failures based on number of landfill failures. Therefore it is most important to understand the effect of compaction and determine the optimum compaction for successful bioreactor/ELR operation. It is important to understand and predict the movement and distribution of fluids within the waste mass. The flow and distribution of water in landfills is complicated because waste is a heterogeneous material consisting of a wide variety of particles with pore sizes and shapes. Solid material types and waste anisotropy (e.g. layers of daily cover) also add complication in predicting or interpreting the flow and distribution of water within landfills. The hydraulic properties of waste will vary with overburden stress and potentially over time through compaction and degradation. Therefore, maintaining the uniform moisture content within the landfill and the understanding the moisture flow become the most critical issue for bioreactor operation. If there is not sufficient infiltration of liquid inside the waste, there will not be uniform moisture distribution of liquid so that the waste remains dry over long period. Therefore it is necessary to estimate the optimum density required for the bioreactor landfill which can ensure necessary infiltration and rapid decomposition of waste. might be a range for optimum density for the municipal solid waste for the waste.

1.3 Objective of the study

The main objective of the current research is to investigate the optimum compaction for bioreactor landfill operation. The certain compaction can maximize the gas generation with sufficient flow of moisture. Therefore, it is necessary to investigate the effect of compaction on the flow behavior and degradation of the MSW and to find the optimum density for the landfilled waste. The current study is focused on finding optimum compaction level for the MSW by considering the hydraulic conductivity, porosity and the rate of methane gas generation at various compacted density/unit weight of the MSW.

The main tasks of the current study are outlined as follows;

1. To determine the physical composition, organic content and moisture content of the waste sample
2. To find out effect of compaction on the hydraulic conductivity, porosity and moisture contents of fresh MSW
3. To investigate the factors affecting hydraulic conductivity of the solid waste
4. To monitor the effect of compaction on the degradation, gas generation, leachate production, pH of leachate and hydraulic conductivity of the solid waste

5. To estimate the optimum compaction level based on the hydraulic properties and degradation of the solid waste

1.4 Organization of dissertation

This dissertation is divided into 6 chapters as summarized below:

Chapter 1 Provides an introduction and presents the problem statement and objectives of the research.

Chapter 2 Presents a literature review of the stages of municipal solid waste (MSW) decomposition, Properties of MSW, hydraulic conductivity, porosity, moisture content and factors affecting hydraulic conductivity, porosity, moisture content, gas generation and degradation of MSW and factors affecting the degradation and gas generation, methods for compacting the waste in the landfill.

Chapter 3 describes the experimental procedures followed to collect samples, to prepare the samples, to build large permeability device, to build laboratory scale landfill reactors. It includes experimental procedures followed to perform hydraulic conductivity test, porosity, moisture content and maximum density of MSW. It also includes experimental procedures followed to monitor degradation of waste.

Chapter 4 Presents the experimental results and discussion on the data obtained through the laboratory tests. Basically the chapter describes the effect of compaction on the hydraulic parameters and degradation of municipal solid waste. Similarly, the chapter includes factors affecting the hydraulic conductivity, the effect of compaction on the waste degradation, gas generation, leachate circulation and generation, pH variation, hydraulic conductivity.

Chapter 5 presents the procedures followed for the estimation of optimum density level of MSW samples. It includes the analysis and discussion to estimate optimum by considering several properties such as hydraulic conductivity, porosities and also gas generation from differently compacted waste in laboratory bioreactor cells.

Chapter 6 summarizes the main conclusions from the current research and provides recommendations for future work

Chapter 2

Literature review

2.1 Back ground of municipal solid waste

According to U.S. Environmental Protection Agency 2014, people of United States of America generated about 251 million tons of trash and recycled and composted almost 87 million tons of this material, equivalent to a 34.5% recycling rate in 2012. On average, USA recycled and composted 1.51 pounds of the individual waste generation of 4.38 pounds per person per day. The generation and recycling rates of waste for 1960 to 2012 has been shown in the Figure 2-1 and Figure 2-2. It is clear that waste generation and recycling is continuously increasing in the United States. Because of increasing generation of waste, the management of waste become important issue.

Figure 2-1 MSW generation rates at different year (USEPA 2014)

Figure 2-2 MSW recycling rates at different year (USEPA 2014)

Municipal Solid Waste is a mixture of wastes that are primarily of residential and commercial origin. Municipal Solid Waste (MSW) is more commonly known as trash or garbage. Typically, MSW consists of everyday items we use and then throw away, such as product packaging, grass clippings, furniture, clothing, bottles, food scraps, newspapers, appliances, paint, and batteries, and soils (both waste products and material used as cover material). This comes from homes, schools, hospitals, and businesses. A wide range of particle sizes is encountered ranging from small soil particles to large objects such as construction and demolition waste, wooden products etc. The proportion of these kinds of materials might be varying from one site to another and also within a site. Specific reasons for regional differences may include (USEPA 2014)

- Variations in climate and local waste management practices
- Differences in the scope of waste streams

- Variance in the per capita generation of some products, such as newspapers and telephone directories, depending upon the average size of the publications
- Variations in economic activity
- Local and state regulations and practices

Life style changes, seasonal factors, pre-treatment and recycling activities result in a changing waste stream over time (Dixon and Jones 2005). It should be noted that the composition of MSW varies from region to region and country to country. The composition of MSW also depends on the economic situation and life style of the people. For example, developing countries often have waste streams that contain more biodegradable material and less non-biodegradable materials such as plastics, and developed countries with well-developed recycling and pretreatment policies have wastes with less biodegradable content and more non-biodegradable. These variations in MSW produce fundamental and significant differences in waste engineering behavior and they must be taken into consideration when using results from the literature.

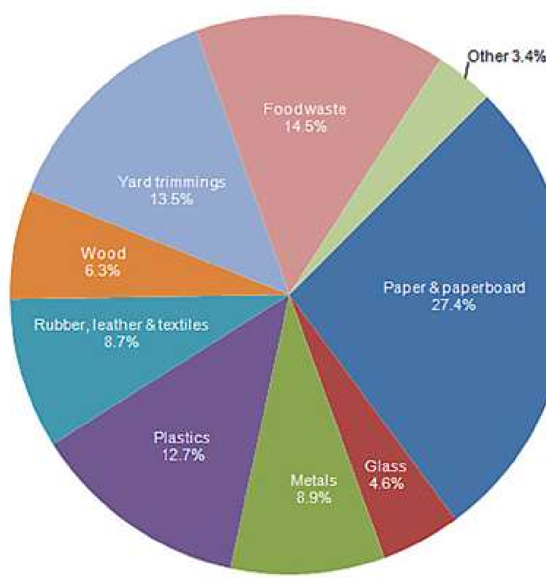


Figure 2-3 Total MSW generation in 2012 (USEPA 2014)

According to Environmental Protection Agency 2014, MSW generated in the USA, organic materials such as paper and paperboard, yard trimmings, and food waste continue to be the largest component of MSW. The Figure 2-3 shows the breakdown of MSW generated, by material. Paper and paperboard account for over 27% and yard trimmings and food waste accounts for another 28%. Plastics comprise about 13%; metals make up 9%; and rubber, leather, and textiles account for almost 9%. Wood follows at over 6% and glass at almost 5%. Other miscellaneous wastes make up approximately 3% of the MSW generated in 2012.According to U.S. Environmental Protection Agency (2014), total MSW recovery in 2012 was almost 87 million tons. The Figure 2-4 describes the recovery rates for different waste constituents across the country. Paper and paperboard account for over 51 % and yard trimmings account for over 22 %, while food waste accounts for another 2%. Metals comprise about 9 %; glass about 4 %; and plastic and wood about 3 % each. Other miscellaneous materials make up about 6 % of MSW recovery in 2012

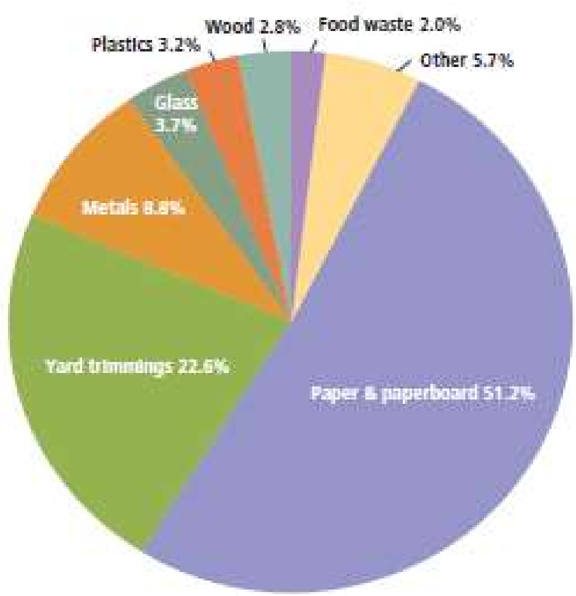


Figure 2-4 Total MSW recovery by material in 2012 (USEPA 2014)

According to USEPA 2014, United States of America also recovered over 65 million tons of MSW through recycling and over 21 million tons through composting in the same year 2012. The recovery of some particular components is given in Figure 2-5.The lead-acid battery recovery was about 96% (2.8 million tons). Newspaper/mechanical papers recovery was about 70% (5.9 million tons), and over 57% of yard trimmings were recovered (19.6 million tons)

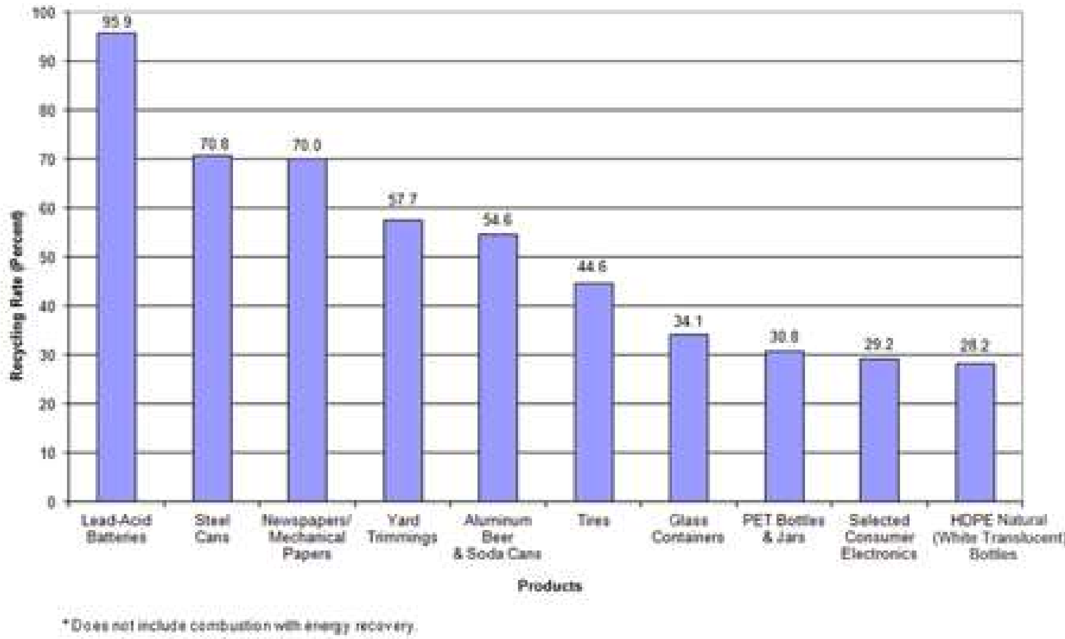


Figure 2-5 MSW recycling rates of selected products in 2012 (USEPA 2014)

Significant amounts of material from each category were recycled or composted in 2012. The highest recovery rates were achieved in paper and paperboard, yard trimmings, and metals. Americans recycled more than 64 % of the paper and paperboard generated. Over 19 million tons of yard trimmings were composted, representing almost a five-fold increase since 1990. Recycling these three materials alone kept over 28 % of MSW generated out of landfills and combustion facilities.

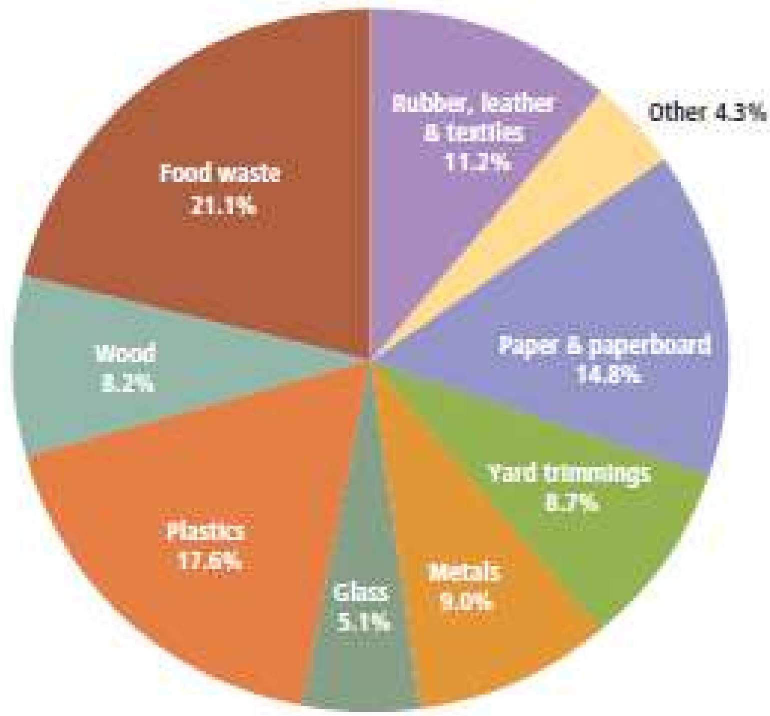


Figure 2-6 Total MSW discards by material in 2012 (USEPA 2014)

It is clear that much of the MSW generation still goes to landfill which is explained by the Figure 2-6. MSW management is one of the big concerns around the world. According to USEPA (2014), 164 million tons waste was discarded into landfill in 2012 after recycling and composting. This is one of the reason bioreactor landfill concepts arise across the country in order to accelerate the process of degradation and to safe land. As degradation starts, the mass of the landfill waste declines, creating more space for dumping waste. Due to addition of water, bioreactor landfills are expected to increase this rate of degradation and it has been reported that there is a 15 to 30 % gain in landfill space when landfilled waste becomes stabilized. As the amounts of solid waste produced is increased every year so that there might be scarcity of landfill spaces in future, bioreactor landfill can thus provide a significant way of maximizing landfill space and minimizing land utilization by the landfill. This is not just cost effective, but since less land is needed for the landfills, this is also better for the environment. Furthermore, the conventional landfills are monitored for at least 3 to 4 decades to ensure that no leachate or landfill gases escape into the community surrounding the landfill site. In contrast, bioreactor landfill are expected to decompose to level that does not require monitoring in less than a decade. But there is not still available of well-developed technique for the landfill design and operation. There are several researches going on in the university and in many other organizations to improve its efficiency. The stabilization, degradation become the serious issue for geotechnical and geo environmental engineers.

2.2 Composition of MSW

A description of the nature and composition of any material can give clues to its likely hydrogeological behavior (Beaven 2000). For example, a material consisting entirely of uniformly graded round particles has well defined void structure i.e. porosity. In this type of materials there is open and well interconnected pore structure leads which leads to a relatively high permeability such as in quartz gravel. The composition of municipal waste is considerably much more complicated than that of uniformly graded gravel. Municipal solid waste consists of a wide range of highly variable materials with a wide particle size distribution. The nature and distribution of pores not only depends on particle shape and size but also much depends on density and on composition of the MSW. According to Beaven (2000), flow can occur between the relatively large voids between individual fragments of waste as well as through the micro-pores of many individual waste fragments (e.g. paper products). As the overall density of the waste increases the macro-pores will tend to collapse resulting in more reliance on flow through the micro-pores or alternatively along the interface between two particles in contact.

Waste composition changes with geographical location, depending on economic conditions, lifestyle, industrial structure and waste management techniques. Guermond et al. (2009) compiled the waste composition information published for various countries (See Table 2-1). It is evident from Table 2-1, that the percent of organic matter in waste is higher in developing nations than developed nations. The waste composition also changes over time due to changes in waste management practices, and the economic development of the region. The change in waste composition found in the U.S. over the last few decades is shown below (See Figure 2-7).

Table 2-1 Waste composition found in different countries (Guermond et al. 2009)

Country	City	Organic matter	Cardboard	Plastics	Metals	Glass
Morocco	Agadir	65-70	18	2-3	5.6	0.5-1
Jordan	Amman	63	11	16	2	2
Turkey	Istanbul	36.2	11.2	3.1	4.6	1.2
Tunisia	Tunis	68	11	7	4	2
Mauritania	Nouakchott	48	6.3	20	4.2	4
Guinea	Labe	69	4.1	22.8(+textile)	1.4	0.3
France	Paris	28.8	25.3	11.1	4.1	13.1
Portugal		35.5	25.9	11.5	2.6	5.4
Greece	Palermo	31.7	23.1	11.8	2.7	8.3
Canada	Toronto	30.2	29.6	20.3	2.1	2

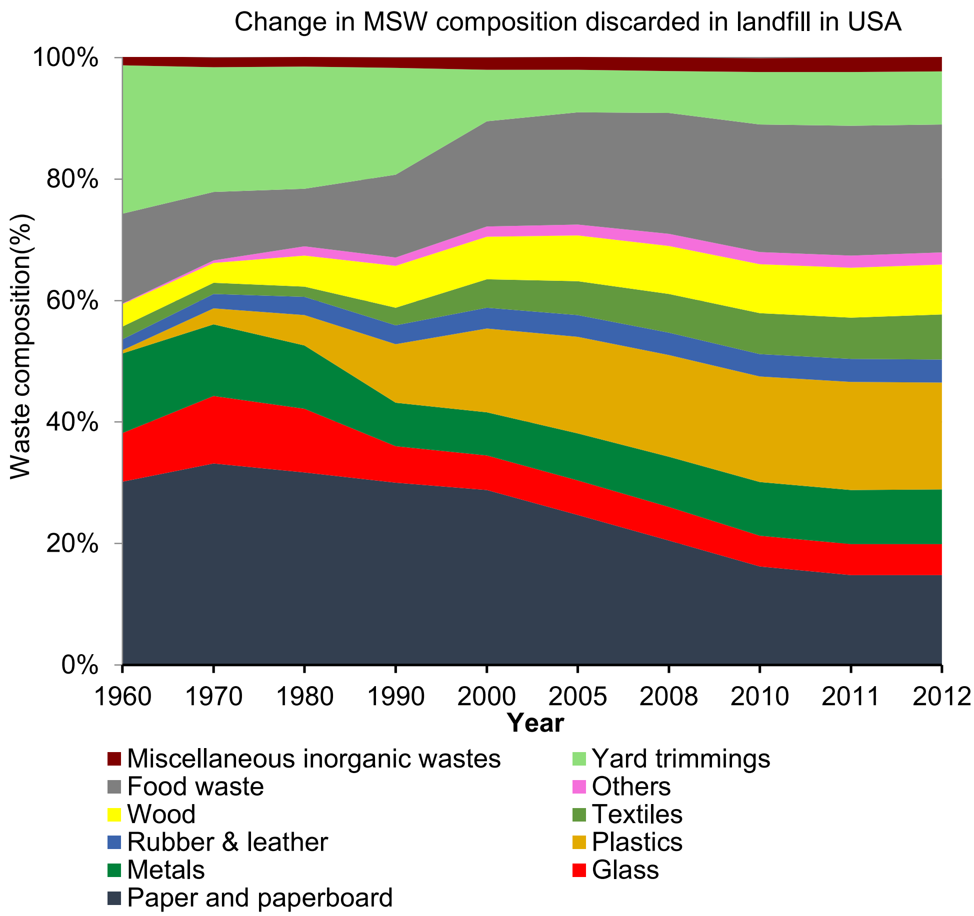


Figure 2-7 Change in waste composition in the USA

Flow is not only affected by composition of the MSW but the amount of methane generated from a landfill depends on the composition of the waste. Further, degradation and gas generation of MSW depends directly on the organic contents in the MSW. Besides this, different types of waste degrade at different rates. Hence, the rate at which methane is generated from landfills also depends on the waste composition.

2.3 Bioreactor landfill/ELR landfill

A landfill is a site for the disposal of waste materials and is the oldest form of waste treatment. Landfills have been the most common method of organized waste disposal and remain so in many places around the world. Landfill disposal is the most commonly used waste management method around the world. The conventional landfills usually include environmental barriers such as landfill liners and covers, which exclude moisture that is essential to waste biodegradation. Consequently, wastes are contained in a “dry tomb” and remain intact for long periods of time ranging from 30 to 200 years, possibly in excess of the life of the landfill barriers and covers.

In recent years, due to the advance in knowledge of waste behavior and decomposition processes, there has been a strong tendency to upgrade existing conventional landfill to a bioreactor landfill. Bioreactor landfills are usually designed to maximize the waste degradation by applying water/liquid into the waste under controlled conditions. The design objectives of these landfills are to minimize leachate migration into the subsurface environment and maximize landfill gas (LFG) generation rates. The bioreactor technology provides control and process optimization, primarily through the addition of leachate or other liquid amendments, the addition of sewage sludge or other amendments, temperature control, and nutrient supplementation. Beyond that, bioreactor landfill operation may involve the addition of air. Based on waste biodegradation mechanisms, different kinds of “bioreactor landfills” including anaerobic bioreactors, aerobic bioreactors, and aerobic-anaerobic (hybrid) bioreactors have been constructed and operated worldwide (Warith et al., 2005) .

Bioreactor landfill is an emerging technology for sustainable solid waste management. The method not only enhances the degradation processes, but also stabilizes the landfill as quickly as possible. Engineered bioreactor landfill sites can reduce the emission of global warming greenhouse gases, and additionally they can provide immediate improvements to the surrounding local environment in terms of controlling odor and methane gas migration (McCreanor et al. 1996; Pohland 1994;

Pohland and Al-Yousfi 1994). This landfill technology is gaining popularity and has been tried in pilot and full scale in various landfills in North America, particularly in areas where landfill closure is costly and/or where landfill space is crucial.

2.3.1 Advantages of bioreactor landfills

Numerous benefits can be derived from the bioreactor landfill operation. These can be in the form of environmental, regulatory, monetary and social benefits. Several researchers (Warith 2002; Pacey et al., 1999; Barlaz et al., 1990; Reinhart and Townsend 1998) had pointed out the numerous advantage of using bioreactor landfill such as:

1. Enhance the gas generation — the operation of bioreactor landfill significantly increases the gas generation and thus the gas can be utilized for energy production.

2. Increased landfill space- by accelerating the degradation resulted in rapid settlement of solid waste. The landfilling capacity increases due to rapid settlement during operational time period

3. Reduce environmental impacts — by recirculation of the leachate and utilizing the LFG emissions, bioreactor landfills will have minimum impact on groundwater, surface water, and the neighboring environment. It also reduces greenhouse gas emissions to the environment.

4. Reduction of leachate treatment and operating cost — a bioreactor landfill enhances the biological and chemical transformation of both organic and inorganic constituents. Because of utilizing generated leachate back within the landfill will reduce the requirement of leachate treatment and thus operating cost.

5. Reduction in post-closure care, maintenance and risk — the operation of bioreactor landfill minimizes environmental risk and liability because of the rapid waste stabilization. Proper operation of a bioreactor landfill will reduce landfill monitoring

activities and post-closure care cost. Landfill operation and maintenance activities are considerably reduced. Landfill monitoring activities can be reduced.

2.3.2 How bioreactor landfill differs from conventional landfill

A conventional landfill is an engineered waste disposal facility where garbage is deposited in the ground compacted into a cell and covered with earth fill materials. Generally wastes are contained in a “dry tomb” and remain intact for long periods of time ranging from 30 to 200 years, possibly in excess of the life of the landfill barriers and covers. The main reason of remaining the waste in non-degraded form is lacking of moisture. Environmental controls are incorporated into the engineering design of the facility to protect both the human and natural environments.

A bioreactor landfill is a sanitary landfill that uses enhanced microbiological technique to transform and stabilize the readily and moderately decomposable organic waste constituents within 5 to 10 years of bioreactor process implementation. The bioreactor landfill operation significantly increases the waste decomposition, conversion rates and process effectiveness. Stabilization means that the environmental performance measurement parameters (landfill gas composition and generation rate and leachate constituent concentrations) remain at steady levels, and should not increase in the event of any partial containment system failures beyond 5 to 10 years of bioreactor process implementation (pacey et *al.,* 1999). The bioreactor landfill requires certain specific management activities and operational modifications to enhance microbial decomposition processes. The single most important and cost-effective method is water/leachate addition and management. Other strategies, including waste shredding, pH adjustment, nutrient addition, waste pre-disposal and post-disposal conditioning, and temperature management, may also serve to optimize the bioreactor process (Pacey et *al.*, 1999).

The bioreactor operation requires significant water/leachate addition to reach and maintain optimal moisture condition. Solid waste has a high moisture holding capacity because of consisting high water absorbent materials. Leachate is usually not available in sufficient quantity to sustain the bioreactor process so that water is added to produce leachate. Basically water is the most are suitable leachate supplement. Although Subtitle D does permit recirculation of leachate and condensate from a specific landfill, many states have not yet endorsed the leachate recirculation option, but only permitted the addition of water or other liquid amendments needed to facilitate the bioreactor activity (Pacey et *al.,* 1999). Shortly following closure of a bioreactor landfill, the landfill gas generation will usually reach maximum within few years and then gradually decline over the next few years to a stable and relatively low and declining rate. Similarly, after landfill closure, many leachate contaminant concentrations will change from levels regarded as highly polluted to much lower levels normally characteristic of extended stabilization (pacey et *al.,* 1999).

2.4 Properties of municipal solid waste

The determination the engineering properties of MSW is the challenging work because of the heterogeneity. Since the municipal solid waste is the combination of different individual waste components, properties of the individual components controls the overall characteristics of the landfill waste. The understanding of the variation in compositional characteristics may be helpful to estimate properties when MSW is subjected to spatial and temporal heterogeneity. Knowledge of the likely ranges of properties of all components is required to assess waste behavior and its properties and hence to design the efficient bioreactor landfills. It is utmost important to have reliable engineering properties of MSW in order to evaluate and predict landfill behavior and hence for landfill operation. However, determining engineering properties is extremely difficult as mentioned by Manasslero et *al.,* (1997) due to the following reasons,

1) Difficulties in sampling of MSW which simulate the in situ condition

2) Lack of generally accepted sampling procedure for geotechnical characterization of waste material

3) Variation in properties of municipal solid waste with time

4) Level of training and education of the personnel on site for basic interpretation and understanding of the measurements, and

5) Heterogeneity of the MSW within the landfill and its variation with geographical location

The properties of MSW are very important in order to design landfill, particularly bioreactor landfill. Generally density, hydraulic conductivity, porosity, compressibility, stiffness and shear strength are most important properties of any solid waste. There are several studies available about the engineering properties of MSW. Unfortunately, due to the lack of both agreed classification system and test standards it is difficult to interpret published results (Dixon and Jones 2005). Generally the characteristic of the waste is not described in detail and the test boundary conditions are also rarely provided. This makes it difficult to apply the results of waste from one particular site other site.

Waste is a highly porous medium with particulate solid material and pore space distributed throughout the mass. The pore space may be filled with liquid and/or gas. The porous medium most closely comparable to solid waste landfills in terms of structure, porosity and gas content is often considered to be unsaturated soil (McDougall et *al.,* 2004). However, a waste is rather more complicated than soil because of the potential for biological and chemical ongoing process and interactions. The solid phase also comprises a wide range of different material types with vastly different mechanical and physical properties which lead to the uncertainty on the waste properties.

2.4.1 Density and unit weight of municipal solid waste

The bulk density (ρ_{wet}) of the solid waste is defined as the total mass of the waste (solid + water) within a unit volume (V_t) of solid waste. The dry density (ρ_{dry}) is defined as the total mass of dry solids within a unit volume of solid waste.

$$\rho_{wet} = \frac{M_s+M_w}{V_t} \quad \text{(2.1)}$$

$$\rho_{dry} = \frac{M_s}{V_t} \quad \text{(2.2)}$$

Similarly unit weight (Y) of MSW is defined as the weight of solid waste per unit volume. It is just equal to the density multiplied by the acceleration due to gravity (g). Like density, MSW has bulk unit weight and dry unit weight respectively.

The unit weight of municipal solid waste (MSW) is an important parameter in engineering analyses of landfill performance, but significant uncertainty currently exists regarding its value (Zekkos et *al.,* 2006).The knowledge of unit weight/density is required in landfill engineering for all aspects in order to design efficient bioreactor landfill. As the landfill is bioreactor, there is recirculation of leachate/water. In order to enhance the degradation, the liquid medium should transport all around the solid waste mass. Permeability is directly connected to the density of the waste so that attention should be given for the density of landfill waste. Density of MSW is also necessary for many engineering analyses of landfill systems, including slope stability, geomembrane puncture, pipe crushing, and landfill capacity evaluation. However, the value of the unit weight of MSW continues to be a major source of uncertainty in landfill performance analyses Significant scatter exists in the reported values of MSW unit weight. Hence, it is difficult for an engineer to estimate with confidence a representative MSW unit weight profile for use in engineering analyses (Zekkos et *al.,* 2006). MSW unit weight varies significantly between sites and within a single site. Since the MSW is heterogeneous materials, it consists of highly variable components, types and amounts of cover soil differ among sites, the percentage of inert and industrial wastes varies placement procedures and environmental conditions, which play an important role in determining the in-situ unit weight (Dixon and Jones 2005). Fassett et *al.,* (1994) considered that the following factors should be recorded along with measured unit weights: MSW composition including daily cover and moisture content; method and degree of compaction; the depth at which the unit weight was measured; and the age of the waste. Significantly different MSW unit weight profiles have been reported in the literature. Hence, it is difficult to estimate a truly representative MSW unit weight for use in waste engineering analyses. The majority of studies did not report the method used to estimate the MSW unit weight. Landva and Clarke (1990) suggested a theoretical approach to measure the density or unit weight of waste. It was recognized that the determination was complicated due to the wide diversity of materials present in the waste and the ability of some components to absorb water. They calculated the possible maximum and minimum densities for a range of waste compositions. A possible range of average unit weights was calculated by considering (1) the lightest combination of materials and their dry unit weight and (2) the heaviest materials and their saturated unit weights. This yielded possible average unit weights of the MSW ranging from 3.8−16.3 kN/m^3. Landva and Clark (1986) also reported the total unit weight near the landfill surface of combinations of refuses and soil cover ranged from 8−17 kN/m^3. Zekkos et al. (2005) reported values of in-situ MSW unit weight varied from 3−20 kN/m^3. It can be concluded that the unit weight of the MSW was varied significantly in the literature which was basically due to the variation in waste composition. MSW is highly heterogeneous material. The factors which influence unit weight are composition of MSW, placement procedure, type and amount of compaction, depth of sampling, moisture content and the thickness of daily cover. The layer thickness and the degree of compaction also influence the unit weight of the MSW.

Similarly, limited information is available 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. Harris (1979) reported an average 7.1 kN/m^3 maximum dry unit weight and 58% optimum moisture content from standard proctor tests on wastes obtained from landfills in England. Harris (1979) indicated that the moisture content of incoming fresh wastes typically varied as a function of weather conditions between 20 and 50% and that water addition during compaction would aid in compaction to obtain a denser fill and for maximum amount of waste placement in a given landfill volume. Reddy et *al.*, (2009 b) conducted standard proctor tests on fresh wastes obtained from a landfill in Illinois and they indicated that the standard effort tests resulted in 420 kg/m^3 maximum dry density and 70% optimum moisture content (see Figure 2-8).

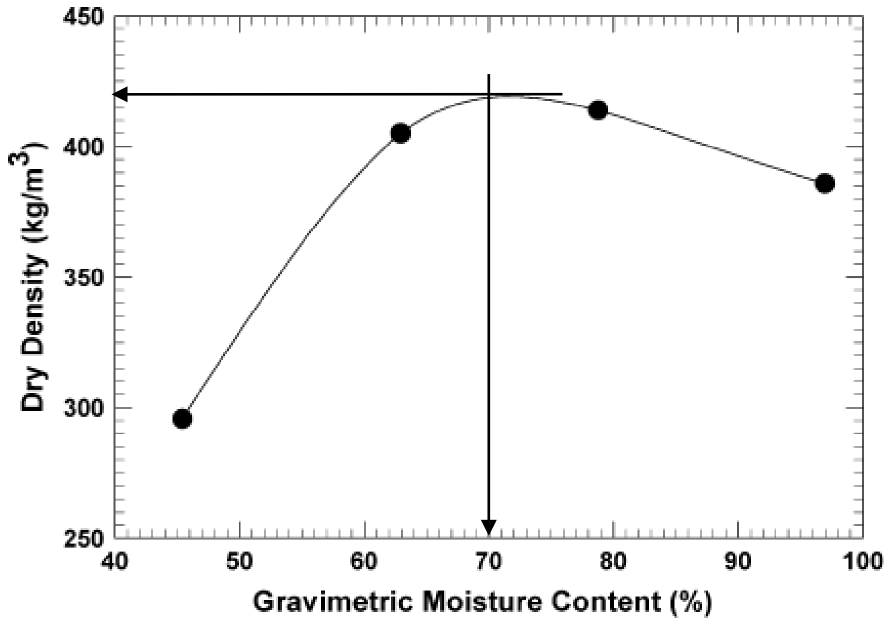


Figure 2-8 Variation of dry density of fresh shredded MSW with gravimetric moisture content (After Reddy et al. 2009 b)

The majority of these tests were conducted using compactive efforts similar to standard effort. 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 allows reducing landfill space requirements to prolong the life of a facility (Ham et *al.,* 1979). Density influences the stability of a landfill. High densities generally associated with high shear strengths and high frictional angle. Combined moisture-density characteristics influence hydraulic response and compressibility of wastes. Overall, the as-placed moisture-density characteristics of MSW are critical for both operation of landfills and engineering response of wastes (Hanson et *al.,* 2010).

2.4.2 Factors affecting unit weight/density of MSW

Unit weight of the waste directly is influenced by age and degree of composition, volume of daily cover, compaction degree, type of waste, total depth of the landfill, and the depth from which the sample is taken (Owis & Kehra, 1990; Chen et *al.,* 2009). There are several factors which might affect the unit weight of waste. Basically, it is affected by compaction effort, the depth of waste (i.e. overburden stress), and the amount of liquid present. Unlike soils, the unit weight of MSW also varies significantly with the large variations in the waste constituents, decomposition and degree of control during placement (such as thickness of daily cover or its absence). It is generally believed that initially the unit weight of waste is much dependent on the moisture content, waste composition, the daily cover soil and the compaction effort during placement. But the unit weight becomes more dependent on the depth of burial, the degree of decomposition and climatic conditions (Dixon and Jones 2005).

2.4.2.1 Waste composition

Since the solid waste is the combination of different components, properties of the individual components controls the overall characteristics of the landfill waste. Individual waste components have a wide range of unit weights and these can change with time. Components may have voids within them in addition to those between components. This results in a significant percentage of waste particles behaving differently to soil particles due to their high compressibility (Dixon and Jones 2005). Several MSW components with organic nature will result in a loss of mass due to degradation process. It is generally believed that degradation results in an increase in waste density, and hence unit weight (Dixon and Jones 2005). Several studies have been conducted to determine the properties of municipal solid waste including unit weight; however, very limited studies focused on the identification of the characteristics of individual components of the solid waste. It is necessary to have knowledge about the characteristics of components in order to better understand MSW properties. Solid waste is extremely heterogeneous material due to presence of various components which lead to the unpredictable on the overall properties of MSW including unit weight. Table 2-2 illustrates the range of densities of components of MSW, together with typical unit weights of the constituents in the dry and saturated conditions which contributes a major role in varying unit weight of MSW.

Table 2-2 Typical refuse composition (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

2.4.2.2 Depth

Oweis and Khera (1990) reported data relating the effect of burial depth on the unit weight of the waste. The results were determined from the waste cores taken from drilling of large diameter on a landfill in Southern California. They made the following conclusion:

1. Unit weight of the waste was increased with the burial depth
2. The dry unit weight of newer and older fill were approximately equal at a given depth

3. The wet weight of more recently placed waste was slightly higher than the wet weight of older waste at an equal depth, largely because of higher water content

The average dry density was calculated approximately 720 kg/m^3 at a depth of 5 m (17 ft) increased to approximately 990 kg/m^3 at a depth of 25 m (80 ft). The corresponding wet densities was varied between 800 kg/m^3 to 930 kg/m^3 at a 5 m and between 1150 kg/m^3 to 1220 kg/m^3 at a depth of 25 m due to moisture variation.

The solid waste is highly compressible in nature unlike soil. Powrie and Beaven (1999) reported the density of waste varies with effective stress, which is a function of depth. They investigated the variation in dry density, saturated density and density at field capacity with vertical effective stress which is shown in Figure 2-9. The data was obtained by compressing samples of waste in a large diameter cylindrical tests chamber. One of the implications of this work, in terms of the waste density achieved, is that compaction at the tipping face can have a similar effect to the burial of the waste by several meters of overburden stress (Powrie et *al.,* 1998. Gourc *et al.,* 2001). The authors reported the bulk unit weight against overburden stress which demonstrated a trend of increasing unit weight with stress level.

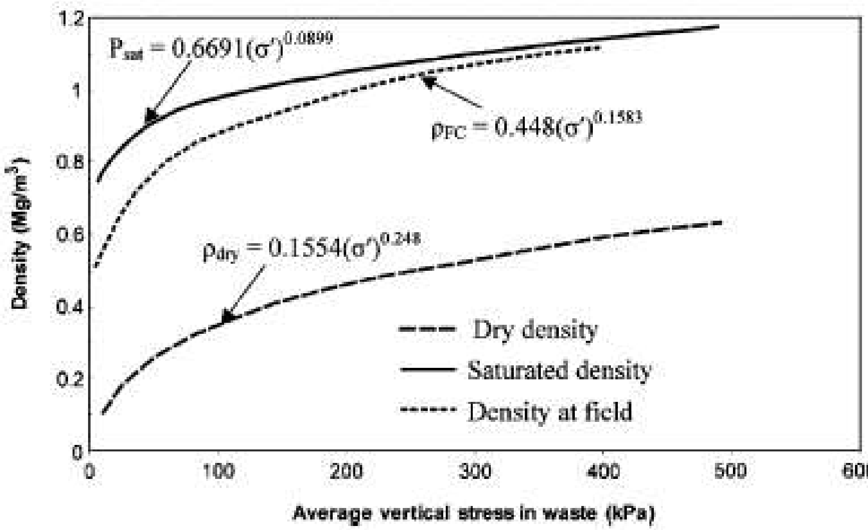


Figure 2-9 Relationships between density and average vertical stress (Powrie and Beaven 1999)

Zekkos et al., 2006 performed a detailed analysis of the MSW unit weight from different landfill sites. All field data show that each individual landfill have a characteristic MSW unit weight profile. A hyperbolic relationship was developed to represent this characteristic MSW unit weight profile based on in situ unit weight data and trends observed in large-scale laboratory tests. Landfill-specific values of MSW unit weight depend primarily on waste composition, operational practices (i.e., compaction, cover soil placement, and liquids management), and confining stress on the context of those characteristic profile. Estimation of the vertical overburden stress requires knowledge of the unit weight of the overburden material and thus an iterative procedure would be required to estimate a MSW unit weight profile. A more convenient equation for MSW unit weight can be written as a function of depth. This form of the relationship is more robust, because field data typically consist of a known unit weight at a known depth and not at a known vertical or mean stress. They recommended hyperbolic MSW unit weight equation has the form,

$$\gamma = \gamma_i + \frac{z}{\alpha+\beta z} \quad (2.3)$$

Where, γ_i = near-surface in-place unit weight (kN/m^3), z = depth (m) at which the MSW unit weight is to be estimated; and α (m^4/kN) and β (m^3/kN) = modelling parameters. Typical values of near-surface in-place unit weight, modelling parameters are provided by Zekkos et *al.,* 2006 (See Table 2-3).

Table 2-3 Hyperbolic parameters for compaction effort and amount of cover soil (Zekkos et. al., 2006)

Compaction effort and soil amount	γ_i (kN/m^3)	β (m^3/kN)	α (m^4/kN)
Low	5	0.1	2
Typical	10	0.2	3
High	15.5	0.9	6

2.4.2.3 Compaction

MSW is a very heterogeneous material and most of its components have a high void ratio and a high compressibility. Thus the compaction processes will reduce the voids within an individual component as well as voids among various components. The unit weight of compacted waste will depend upon the waste components, thickness of layer, weight and type of compaction plant and the number of times equipment passes over the waste (Dixon and Jones 2005). Fassett et *al.,* (1994) reported a detailed survey of bulk unit weight data obtained from the international literature. A statistical analysis of the data is shown in Table 2-4. As the compaction process is proving an important parameter for controlling unit weight of MSW, it might result the large variation in unit weight with little or no compaction is used. Dixon and Jones (2005) indicated that current practice is only achieving 'moderate' levels of compaction as per Fassett et *al.,* 1994 compaction criteria for placement of fresh MSW.

Table 2-4 Statistical summaries of bulk unit weight for fresh municipal solid waste (Fassett et al. 1994)

Label of compaction	Poor comp.	Moderate comp.	Good comp.
Range (kN/m^3)	3.0–9.0	5.0–7.8	8.8–10.5
Average (kN/m^3)	5.3	7.0	9.6
Standard deviation (kN/m^3)	2.5	0.5	0.8
Coefficient of variation (%)	48	8	8

Ham et *al.,* (1979) performed detailed tests on the density of milled and unprocessed waste in a laboratory and fields. They reported wet weight density of unprocessed waste increased from 800 kg/m^3 to 950 kg/m^3 at applied stress from 400 to 830 kPa at constant moisture content within the waste. Higher waste density could also be achieved by increasing the magnitude of the applied vibratory force at a same stress.

2.4.2.4 Moisture content

Unit weight of MSW depends on the moisture content of the waste. Compaction effort is also greatly affected by the initial moisture content of fresh waste. The proper moisture content in the fresh waste might reduce the energy required for compaction. It might also affect the workability while doing the compaction of the fresh MSW. Harris (1979) reported an average 7.1 kN/m3 maximum dry unit weight with 58% optimum moisture content and Reddy et al. (2009 b) obtained 4.1 kN/m3 maximum dry unit weight and 70% optimum moisture content from the standard proctor tests. The optimum moisture content may vary with waste composition. The standard proctor result indicates that waste can get a maximum dry unit weight at specific optimum moisture content in waste. Moisture content of waste depends on the initial waste composition, local climatic conditions, operating conditions and organic content. Some components of waste have high water absorbing capacity which increases individual particle unit weight and thus results in the increase in bulk unit. The addition of liquid and re-circulation of leachate will both have a fundamental influence on moisture contents, and hence on the bulk unit weight of waste

2.4.3 Hydraulic conductivity

Hydraulic conductivity is a property of material which permits the passage of any fluid through its interconnecting pores. The parameters that influence the hydraulic conductivity of MSW are compaction effort, density, particle size, porosity, composition, degree of saturation, and depth within the landfill. According to USEPA regulations, the leachate head over the bottom liner should not be exceeded by 0.3 m. Therefore, a leachate collection and removal system (LCRS) is designed to remove leachate. The hydraulic conductivity of municipal solid waste (MSW) must be estimated for the design of the landfill containment systems (Sharma and Reddy 2004). The ability to transport leachate/water around a landfill mass, either to enhance biodegradation by increasing the water content of the waste or as a means to flush soluble pollutants from the landfill is crucial to the design of a sustainable landfill (Beaven 2000). Waste hydraulic conductivity is important to landfill designers because of the influence it has on leachate pressure distributions in the waste body and hence on the magnitude and distribution of effective stresses and therefore on shear strength (Dixon and Jones 2005). Because of this the hydraulic conductivity of the waste is the single most important parameter for the operation of bioreactor landfill operation. Most of the properties of the MSW are greatly influenced by compaction level. Compaction effort leads to increase the density and lower the porosity which is considered to be the most important parameter of flow within landfill. Powrie and Beaven (1999) performed hydraulic conductivity tests on various wastes using a large size compression chamber. They found that the hydraulic conductivity of non-degraded MSW could reduce by over three orders of magnitude to approximately 10^{-8} m/s between placement and burial to a depth of 60m due to compression. The stress dependency of waste hydraulic conductivity has major implications for the operation of leachate extraction and recirculation systems, and basal and side slope drainage design (Dixon and Jones 2005).

In addition, the hydraulic conductivity of MSW varies spatially and with time depending on the extent of the degradation. In general, the hydraulic conductivity of any porous media is primarily a function of the interconnected pore named as porosity. In the case of soils, correlations have been observed between hydraulic conductivity and the porosity. However, MSW is heterogeneous materials which is undergoing on continuous degradation process and there is also mass reduction with time. Solid mass of MSW is a function of time and hence void ratio may not be the best parameter to explain the void space in MSW hence porosity may not be the best parameter to explain the hydraulic conductivity in MSW (Reddy et *al.,* 2009 a). MSW generates gas due to degradation so that flow can fluctuate significantly with time. The presence of landfill gas within the waste will interact with and affect water flow; and both saturated and unsaturated flow occurs (Beaven et *al.,* 2011). In order to deal with duel state of waste mass, it is important to consider both saturated and unsaturated hydraulic conductivity.

2.4.3.1 Saturated hydraulic conductivity

The hydraulic conductivity of saturated waste materials is assumed to obey Darcy's law: the rate of liquid flow (Q) through a unit cross sectional area (*a*) under unit hydraulic gradient (*i*) is proportional to the hydraulic conductivity (*k*) of the materials.

$$Q = k \times i \times a \ldots\ldots\ldots\ldots\ldots\ldots\ldots \text{(2.4)}$$

The hydraulic conductivity of MSW has been reported from a number of laboratory, and full-scale field studies (Fungaroli and Stiner 1979; Korfiatis et al., 1984; Oweis et al., 1990; Bleiker et al.,1993; Shank 1993; Zeiss and Major 1993; Townsend et al., 1995; Landva et al., 1998; Powrie and Beaven 1999; Jain et al., 2002; Wysocki et al., 2003; Jain et al., 2006; Reddy et al., 2009 a). Most of the laboratory hydraulic conductivity tests were performed by saturating the waste sample. Also most tests, especially in the laboratory, have determined hydraulic conductivity in vertical flow whereas in a landfill the anisotropy resulting from the daily cover soil and layering of the waste would be expected to result in preferential horizontal flow. The determination of the horizontal hydraulic conductivity is quite complex in laboratory. Hudson et al., 2009 reported higher horizontal than vertical hydraulic conductivities on various waste using numerical modeling MODFLOW, and there is a general trend of an increase in k_h:k_v ratios. Landva et al., (1984) performed permeability tests within a 470 mm diameter consolidometer. A variety of materials from different landfills were tested under applied stress up to 400 kPa. Hydraulic conductivity varied from 6.8×10^{-5} m/s at stress of 20 kPa to 6×10^{-9} m/s at stress of 400 kPa. Bleiker (1993) determined the hydraulic conductivity of waste obtained from varying depth within a landfill. The hydraulic conductivity of the waste varied between approximately $1\text{X}10^{-6}$ and 5×10^{-9} m/s for dry densities between approximately 500 and 1200 kg/m3. Powrie and Beaven (1999) performed constant head flow test on crude unprocessed household in a large scale compression cell concluded that the coefficient of permeability decreases with the increase in the effective stress from 10^{-3} m/s to 10^{-7} m/s, when the stress increases from 50kPa to 850 kPa respectively. Powrie et al., (2005) indicated decrease in permeability with increase in effective stress, density and decrease in porosity. Reddy et al., (2009 a) studied the variation in the hydraulic conductivity for fresh MSW ranged from 0.2 cm/s for 4.1 kN/m^3 dry unit weight (under zero vertical stress) and then decreased to 4.9×10^{-5} cm/ s for 13.3 kN/m^3 dry unit weight (under the maximum applied normal stress of 276 kPa). The hydraulic conductivity of the landfilled MSW decreased from 0.2 cm/s to 7.8×10^{-5} cm/s when the dry unit weight increased from 3.2 to 9.6 kN/m^3. The results clearly demonstrated that the hydraulic conductivity of MSW can be significantly influenced by vertical stress and it is mainly attributed to the increase in density leading to low void ratio. In addition to stress level the hydraulic conductivity is also varied with time. Chen and Chynoweth (1995) studied the variation in permeability with density of the samples and indicated a decrease in permeability with time. They performed constant head tests at densities 160 kg/m^3, 320 kg/m^3, and 480 kg/m^3. The permeability reduced from 9.6×10^{-2} cm/s to 4.7×10^{-5} cm/s when compacted from 160 kg/m^3 to 480 kg/m^3 dry density. As well as the variation with density, hydraulic conductivities varied with time and hydraulic gradient. This was attributed to changing pore geometry over time and possibly gas generation. Several studies are available on saturated hydraulic conductivity performed in the laboratories which are summarized in Table 2-5.

Table 2-5 Hydraulic conductivity of MSW based on laboratory Studies

Sources	Unit	Hydraulic Conductivity(cm/s)
Fungaroli and Steiner (1979)	NA	10^{-2} to 10^{-4}
Krofiatis et al., (1984)	8.6 kN/m^3	5.0×10^{-3}–3.0×10^{-3}
Noble and Arnold (1991)	NA	8.4×10^{-4} – 6.6×10^{-5}
Blieker et al.(1993)	5.9–11.8 kN/m^3	1.6×10^{-4}–1.0×10^{-6}
Brandl(1994)	9.0-17.0 kN/m^3	2.0×10^{-3}–3.0×10^{-3}
Chen and Chynoweth(1995)	1.57–4.71 kN/m^3	9.6×10^{-2} –4.7×10^{-5}
Gabr and Valero (1995)	7.4–8.2 kN/m^3	1.0×10^{-3}–1.0×10^{-5}
Beaven an Powrie(1995)	5–13 kN/m^3	1.0×10^{-2}–1.0×10^{-5}
Powrie and Beaven (1999)	3.8 kN/m^3	1.5×10^{-4}–3.4×10^{-5}
	7.1 kN/m^3	2.7×10^{-6}–3.7×10^{-8}
Jang et al., (2002)	7.8–11.8 kN/m^3	1.1×10^{-3}–2.9×10^{-4}
Durmusoglu et. al.(2006)	123–369 kPa	1.2×10^{-2}– 4.7×10^{-4}
Oliver and Gourc (2007)	NA	1.0×10^{-2}–1.0×10^{-4}
Penmethsa (2007)	6.4–9.3 kN/m^3	1.0×10^{-2}–8.0×10^{-4}
Reddy et al (2009 a)(a) Rigid wall permeater test (b) Triaxial Permeater test (c) Variation with Pressure	3–13 kN/m^3	1.0×10^{-1}–1.0×10^{-3}
	4–12 kN/m^3	1.0×10^{-3}–1.0×10^{-6}
	69–276 (kPa)	10^{-4} –10^{-6}
Staub et al (2009)	3.63–5.69 kN/m^3	1.0×10^{-3}–7.5×10^{-3}

It is most important to understand the effect of compaction level on the flow pattern for successful bioreactor operation. The flow pattern within the waste will widely vary as a result of anisotropy, heterogeneity, partial saturation, presence of landfill gas, variation in waste density and effective overburden stress within the landfill (Beaven et *al.,* 2011). However, there is a limitation in previous studies to find the optimum compaction level that is required to maintain uniform flow without hindering the degradation and stability of MSW. There is limited information in the literature on the effect of compaction on the hydraulic conductivity for the required level of compaction of waste and therefore the current understanding is incomplete. Since the compaction is the major parameter influencing the flow inside the waste mass, it must be dealt in various ways to the waste.

2.4.3.2 Unsaturated hydraulic conductivity

The hydraulic conductivity of any unsaturated materials, or soil, will be less than that of same saturated materials, or soil. The main causes of having lesser value of hydraulic conductivity are as (Hillel 1971):

1. Some pores fill with air, reducing the cross sectional area through which flow occur
2. The larger pores empty first so that flow is restricted to smaller pores
3. The tortuosity of the flow path through the interlinked pores increases

When the material is in unsaturated state, water is held in pores by the surface tension forces and by the physical attraction of water to the soil particle interfaces. These forces result in a negative water pressure head or suction head. The volumetric water content (θ) is related to suction head. Flow will occur from areas of high pressure to low pressure head. Water content and hydraulic conductivity will vary at different points along the flow path. Therefore the hydraulic conductivity is both a function of water content and suction head.

Most of the waste in a landfill will be above the leachate level and therefore there will be flow in unsaturated condition. Most of the laboratory based researches have been focused on the hydraulic conductivity in saturated condition and very few studies are available on unsaturated state which may be due to more complex and difficult to run the test in unsaturated condition. As only bottom part of the landfill is only in saturated condition and also the presence of landfill gas would tend to reduce the degree of saturation, even at the base (Beaven et al., 2011) so that it is appropriate to use the concept of unsaturated while in designing the landfill. However, saturated waste hydraulics is crucial as it provides a stepping stone towards understanding unsaturated flow.

Generally, the landfilled waste is not fully saturated and the degree of saturation may vary within the landfill mass spatially and temporally. The degree of saturation and moisture content of the waste is expected to affect the hydraulic conductivity so that it is necessary to have an idea about the unsaturated hydraulic conductivity of the waste. The hydraulic conductivity of a porous medium varies significantly with the degree of saturation (Fredlund and Rahardjo 1993), and this is true in wastes as in soils. According to Radcliffe & Šimůnek (2010), the hydraulic conductivity decreases several orders of magnitude as the soil becomes unsaturated and the decrease for a coarse textured soil is so dramatic that eventually the hydraulic conductivity becomes smaller than that for loam and clay. As municipal solid waste is a coarse textured material, the effect of degree of saturation is expected to be more.

2.4.4 Factors affecting hydraulic conductivity

The hydraulic conductivity of MSW depends on density, particle size, porosity, waste composition, degree of saturation, stage of decomposition, and depth within the landfill. Waste density depends on the composition, compaction process, surface cover, over burden pressure and moisture content of the waste during compaction. In addition to the particle size, the application of daily and intermediate soil covers leads to anisotropy and heterogeneity within a landfill (Powrie and Beaven 1999; Hyder and Khire 2004). Hence, along with the physical properties of the solid waste, various other factors like waste deposition and amount of compaction, type of cover soil, depth of landfill contribute to the variation in hydraulic conductivity and on the behavior of MSW in a landfill.

2.4.4.1 Stress and density

The relationships between hydraulic conductivity and the dry density of waste appear to be well defined but they are different for individual waste types. Many authors have presented data relating hydraulic conductivity to waste density and stress. An example of this is given by Landva and Clark (1986, 1990), who undertook large-scale percolation tests in pits excavated at the surface of various landfills in Canada. The hydraulic conductivity was estimated on the basis of the rate of fall of the water level and flow nets applicable to each stage. Hydraulic conductivities ranging between 4×10^{-4} m/s and 1×10^{-5} m/s were reported. The unit weights of the refuse excavated from the pits generally fell in the range 10 to 14 kN/m^3. However, there was poor correlation between hydraulic conductivity and refuse density, almost certainly due to variations in material composition between the different sites - a small amount of cover material mixed into a predominantly MSW matrix would affect the waste density considerably. Other authors have taken a systematic approach to investigating the relationship between waste density and the logarithm of the hydraulic conductivity. Beaven 2000 and Hudson et *al.,* 2001 reported hydraulic conductivity tests on various samples in Pitsea compression cell. They studied the effect of density, stress and drainable porosity on the hydraulic conductivity. According to Beaven (2000) and Hudson et *al.,* (2001), (1) there was a single correlation for all samples between the logarithm of the vertical hydraulic conductivity and vertical effective stress in first loading (see Figure 2-10). Differences in hydraulic conductivity resulting from particle size reduction and waste degradation are essentially second order, but appear to become more significant at higher vertical effective stresses (with a spread of just over one order of magnitude in hydraulic conductivity at a vertical effective stress of 500 kPa).(2) There are individual correlations between the logarithm of the vertical hydraulic conductivity and density for each waste type, with an essentially linear relationship between the logarithm of the vertical hydraulic conductivity and the dry density (see Figure 2-11)

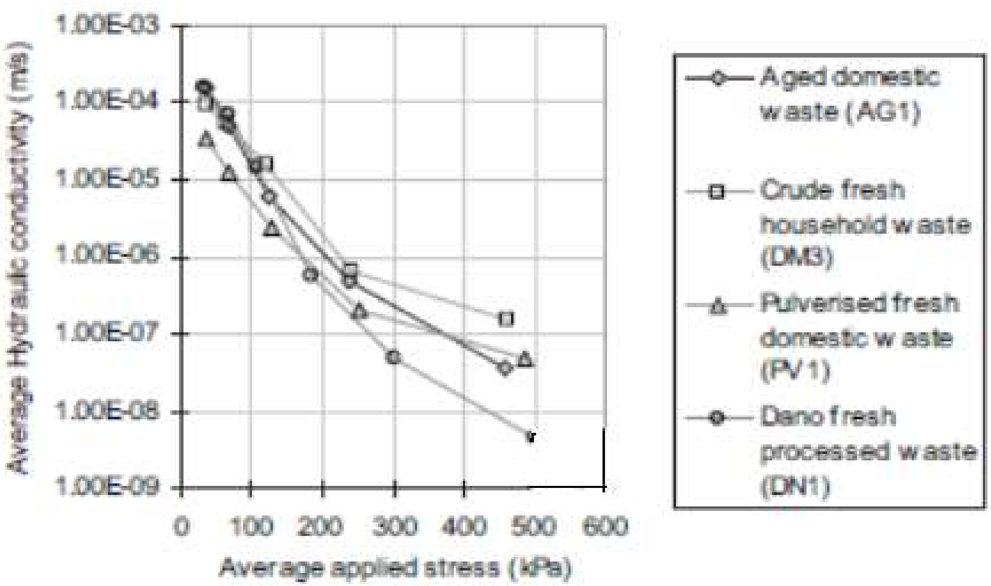


Figure 2-10 Vertical hydraulic conductivity against the logarithm of the vertical effective stress in first loading (Beaven 2000 and Hudson et al. 2001)

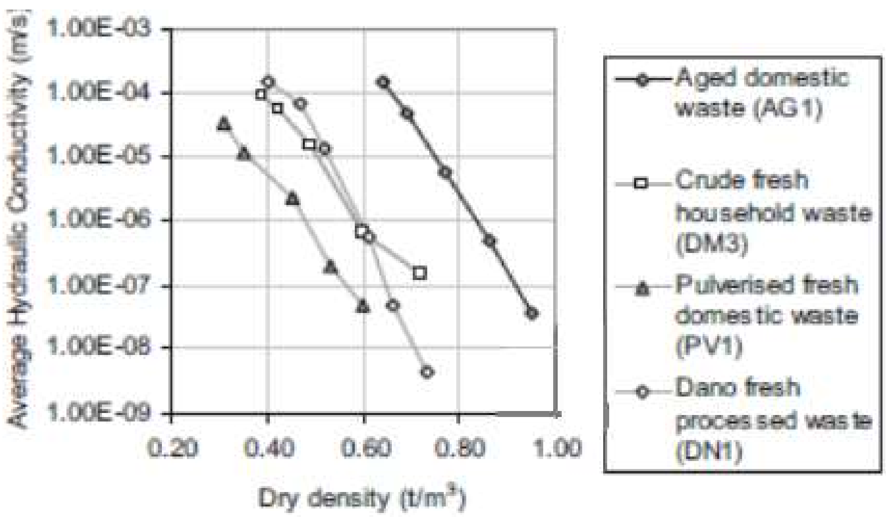


Figure 2-11 Vertical hydraulic conductivity against density for four waste types (Beaven 2000 and Hudson et al. 2001)

Reddy et al. (2009 a) carried out hydraulic conductivity tests on fresh and landfilled MSW using a small-scale rigid-wall permeameter. They reported a range of hydraulic conductivity of 2.8×10^{-3}–11.8×10^{-3} cm/s for the fresh MSW. The dry unit weight of these samples varied in a narrow range between 3.9–5.1 kN/m^3. Similarly, the landfilled MSW tested using the same permeameter produced results between 0.6×10^{-3} and 3.0×10^{-3} cm/s for 4.5–5.5 kN/m^3 dry unit weights. No trend was observed between the hydraulic conductivity and either dry unit weight or age of MSW under the tested conditions in their study. They also performed hydraulic conductivity tests on the large scale rigid wall permeameter. The hydraulic conductivity obtained from large-scale rigid wall permeameter tests is shown in Figure 2-12. The hydraulic conductivity was decreased with increase in normal stress for both fresh and landfilled MSW.

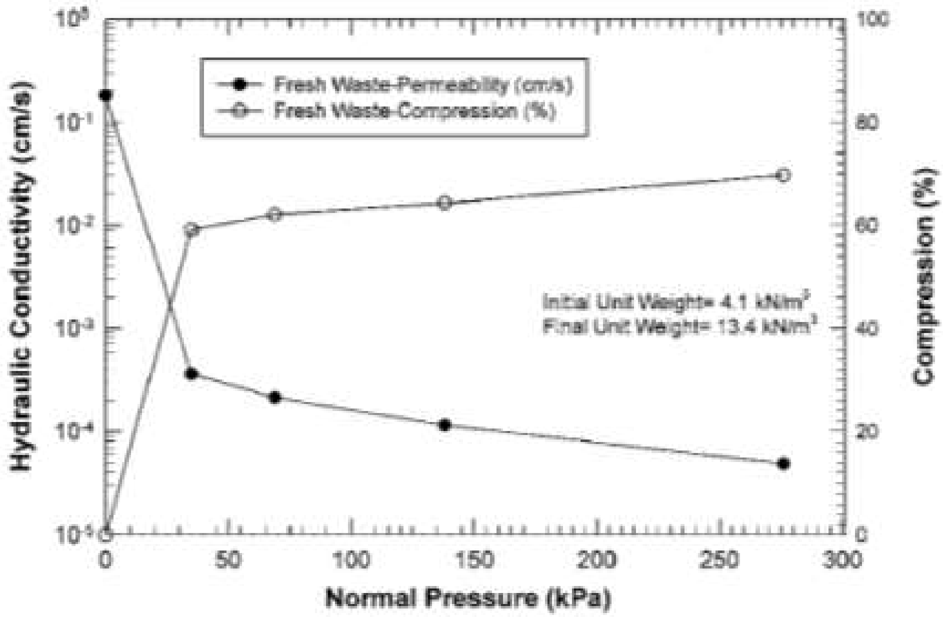
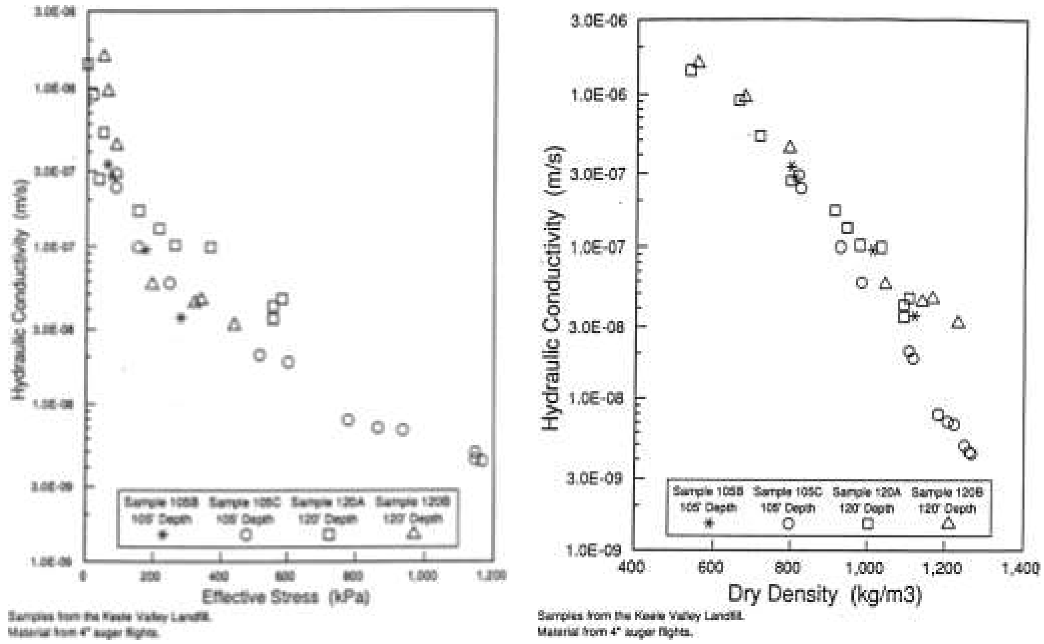


Figure 2-12 Variation of hydraulic conductivity and compression in large scale rigid wall permeameter with normal pressure on fresh MSW (Reddy et al. 2009 a)

Bleiker et al. (1993) performed an extensive series of laboratory based tests on the hydrogeological properties of the disturbed samples of solid waste, obtained from the drilling of a well at Keele landfill in Toronto, Ontario. The materials were compressed with applied stresses up to 1200 kPa and data were collected from resulting density, porosity and hydraulic conductivity of the samples. The hydraulic conductivity decreased with increasing applied stress and dry density (Figure 2-13 a, b). The changes in hydraulic conductivity were over several orders of magnitude and suggest that significant changes in refuse hydraulic conductivity between top and bottom of landfill are possible. They did not consider the effect of biological and chemical clogging in their study which would occur over time in a landfill environment and would further reduce hydraulic conductivity.



(a) (b)

Figure 2-13 (a) Hydraulic conductivity versus effective stress (b) Hydraulic conductivity versus dry density (Bleiker 1993)

2.4.4.2 Depth

Rowe and Nadarajah (1996) reported hydraulic conductivity data for the fresh Kills landfill, New York. The range of hydraulic conductivity values spanned 6 orders of magnitude from 10^{-3} to almost 10^{-9} m/s. The following empirical relationship (equation 2.5) between hydraulic conductivity (k in m/s) and depth (z in meter) was proposed.

$$k = 0.00018 \times e^{-0.269 \times Z} \quad (2.5)$$

The waste is compressible material so that the overburden stress is increased with the depth. The hydraulic conductivity of the waste decreased with depth, the likely result of greater overburden pressures associated with deep locations of the landfill (Jain et al. 2006). Landva et *al.,* 1998 compiled data of their own study with previous study performed by Landva and Clark (1984) and Rowe and Nadarajah (1996) which is shown in Figure 2-14. The plot shows considerable variation in the hydraulic conductivities at each depth. They used a unit weight of $10 kN/m^3$ to convert to an equivalent depth.

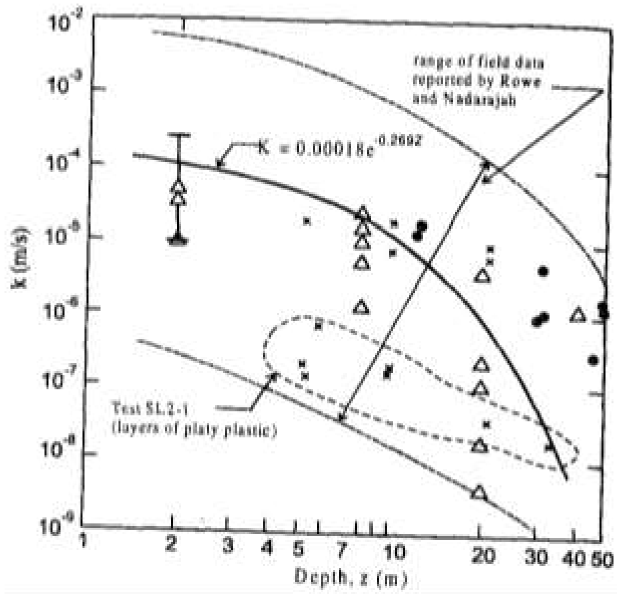


Figure 2-14 Comparison of laboratory and field measured permeability on refuse (After Landva et al. 1988)

2.4.4.3 Porosity

There are not many studies available about the direct relationship between porosity and hydraulic conductivity in the MSW. Some researchers use indirect method to connect between hydraulic conductivity with porosity from dry density. As the dry density increases, the porosity decrease, which causes the vertical saturated hydraulic conductivity to decrease (Staub et al. 2009). Staub et *al.,* 2009 performed the test to measure hydraulic conductivity, open porosity and effective drainable porosity at different dry densities (Figure 2-15). They found that the major driver for hydraulic conductivity is the effective porosity. However the covered density range is small and the number of tests performed is not sufficient to provide for a general assessment of the influence of porosity on the permeability.

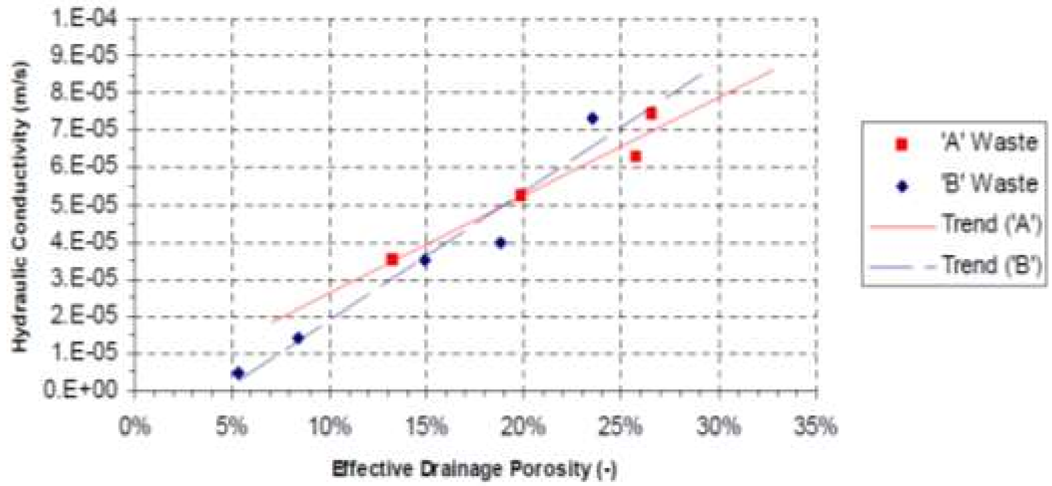


Figure 2-15 Correlation between effective porosity and hydraulic conductivity (Staub et al. 2009)

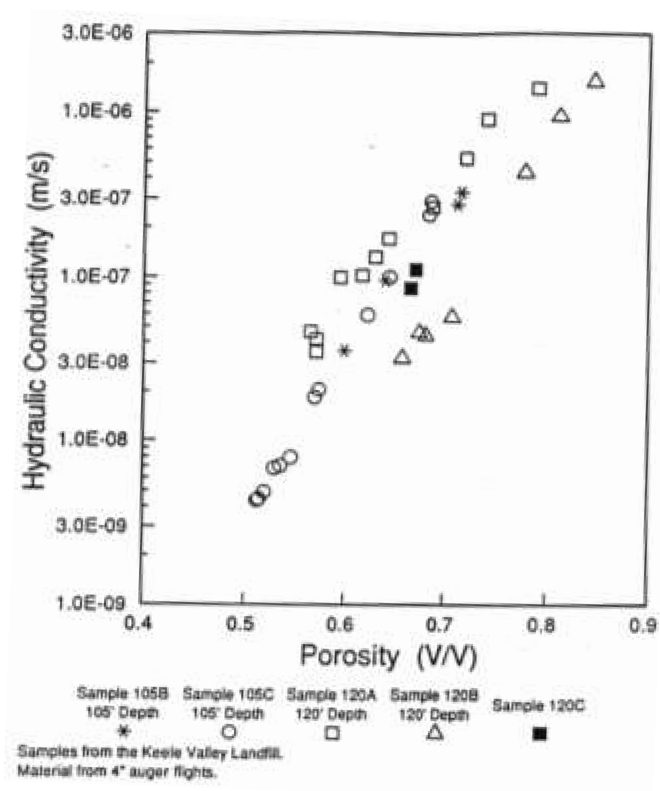


Figure 2-16 Hydraulic conductivity versus porosity (Bleiker 1993)

According to tests performed by Bleiker et *al.,* (1993), the porosity was not related to waste density, but was presented graphically as a relationship with hydraulic conductivity (Figure 2-16). Beaven 2000 and Hudson et al., 2001 reported hydraulic conductivity tests on various samples in Pitsea compression cell. They studied the effect Figure 2-16of density, stress and drainable porosity on the hydraulic conductivity. According to and Hudson et *al.*, (2001) there is a single correlation between the logarithm of the vertical hydraulic conductivity and the drainable porosity of the waste (Figure 2-17). The drainable porosity represents a measure of the size and degree of connectivity of the voids, both of which will have a major influence on the bulk hydraulic conductivity.

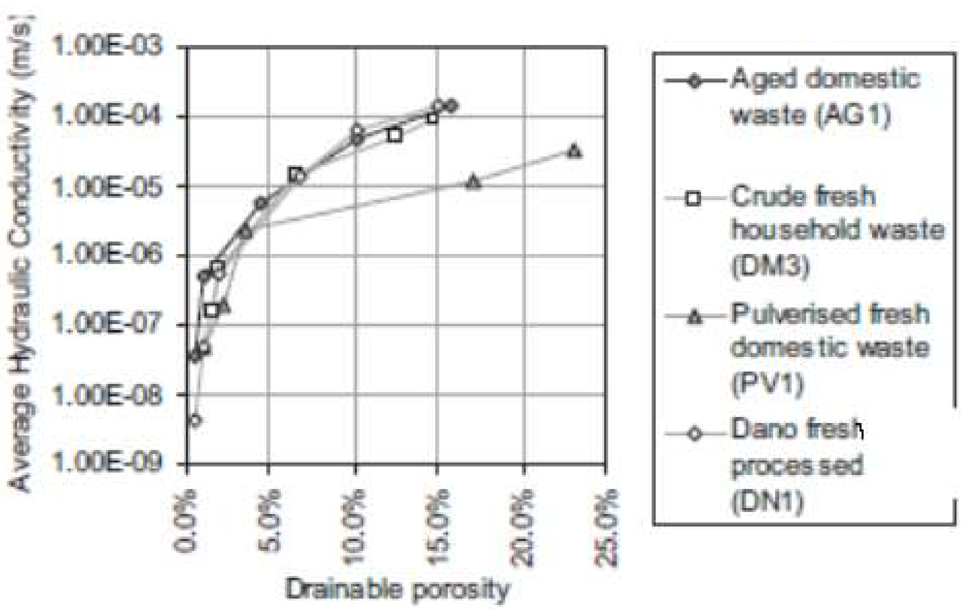


Figure 2-17 Vertical hydraulic conductivity against the drainable porosity (Beaven et al. 2011)

2.4.4.4 Time

Waste is a material which is going continuous degradation process so that mass reduction phenomena will occur along with generation of gas. The mass of the waste converts to gas with the bio degradation process. Hydraulic conductivity of MSW changed over time unlike the soil. Chen and Chynoweth 1995 conducted hydraulic conductivity tests on dry municipal solid waste (MSW) samples compacted in Plexiglas columns to densities of 160, 320 and 480 kg/m^3. The columns were set-up as constant head permeameter and water was flowed continuously through the columns. They found the hydraulic conductivity was time-dependent (Figure 2-18). The temporal variation was attributed to varying degrees of saturation due to gas formation and relative movement of fine particles in the columns. The average hydraulic conductivities at 160, 320 and 480 kg m^{-3} were $9{\cdot}6\times10^{-2}$, $7{\cdot}3\times10^{-4}$ and $4{\cdot}7\times10^{-5}$ cm s^{-1}, respectively.

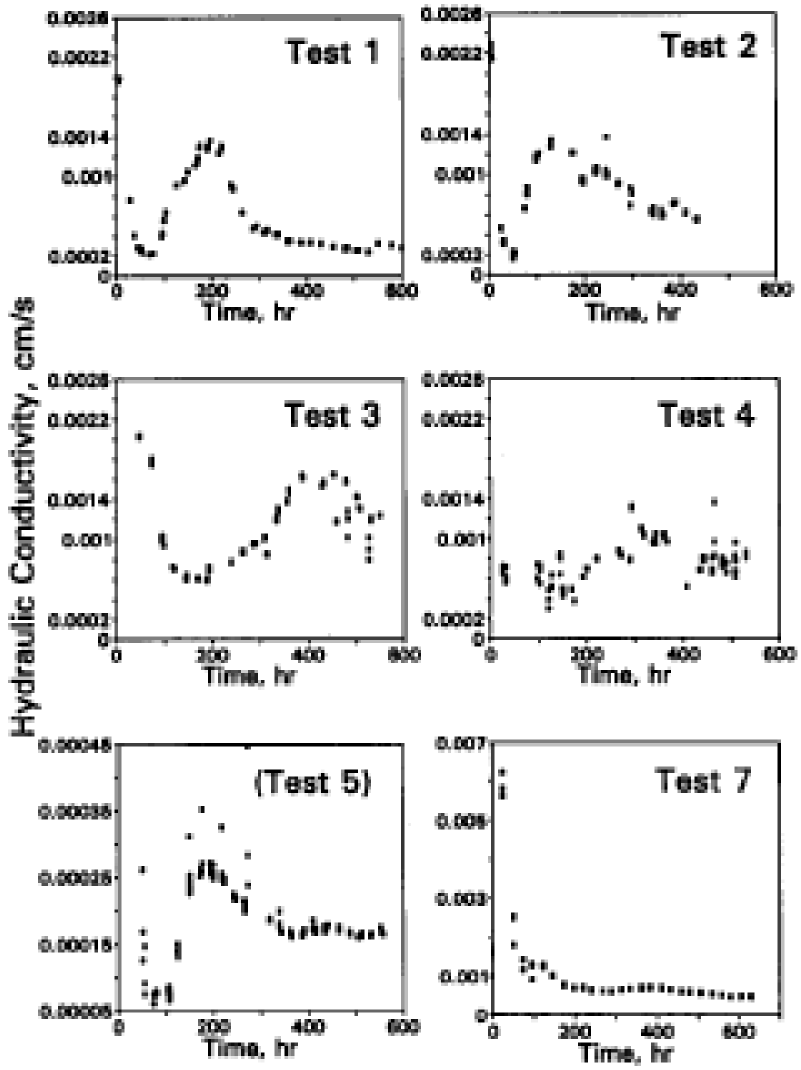


Figure 2-18 Temporal profiles of hydraulic conductivity at densities of 320kgm[3]. Tests 1 & 2 used RDF, 3 &4 used RDF-YW, & 7 used paper as the test material (Chen and Chynoweth 1995).

2.4.4.5 Waste anisotropy

Generally anisotropy refers to the variation of horizontal hydraulic conductivity to the vertical hydraulic conductivity. The literature review reveals a very wide variation in the ratio of horizontal to vertical hydraulic conductivity. Hudson et *al.,* (2009) discuss the impact of anisotropy on the hydraulic conductivity of MSW-type materials. They performed the tests on samples of a 20 year old waste recovered from a landfill (AG2) and a fresh, Dano-processed waste (DN1). The determination of horizontal hydraulic conductivity from flow tests was analytically quite complex and evaluated by numerical modelling using MODFLOW. The result for fresh Dano processed (DN1) as shown in Figure 2-19 in terms of the ratio k_h:k_v as a function of applied stress.

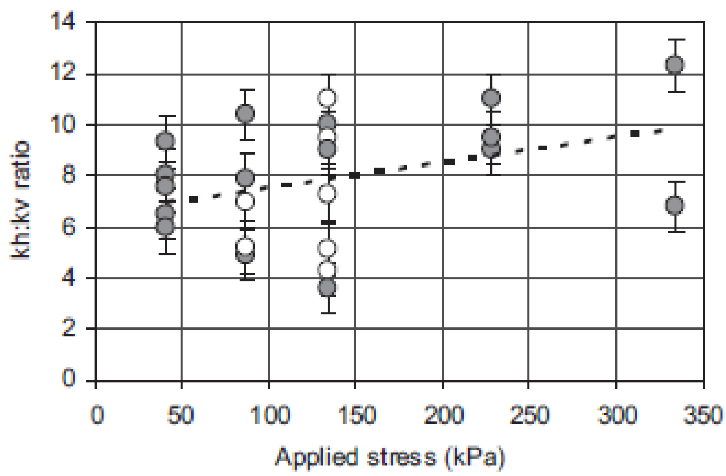


Figure 2-19 Variation of kh: kv to the applied stress for sample DN1 (Hudson et al. 2009, Beaven et al. 2011)

2.4.4.6 Waste degradation

Hossain et al. (2009) performed a series of constant head permeability tests on the samples generated in laboratory scale bioreactor landfills to determine the effect of degradation on the permeability of MSW. They reported that the permeability of MSW in bioreactor landfills decreases with decomposition which is due to decrease in porosity (Figure 2-20). According to their research, the permeability of MSW at first phase of degradation was determined as 8.8×10^{-3} cm/s at density of 700 kg/m^3 and it decreased to 1.3×10^{-3} cm/s at the same density at fourth phase.

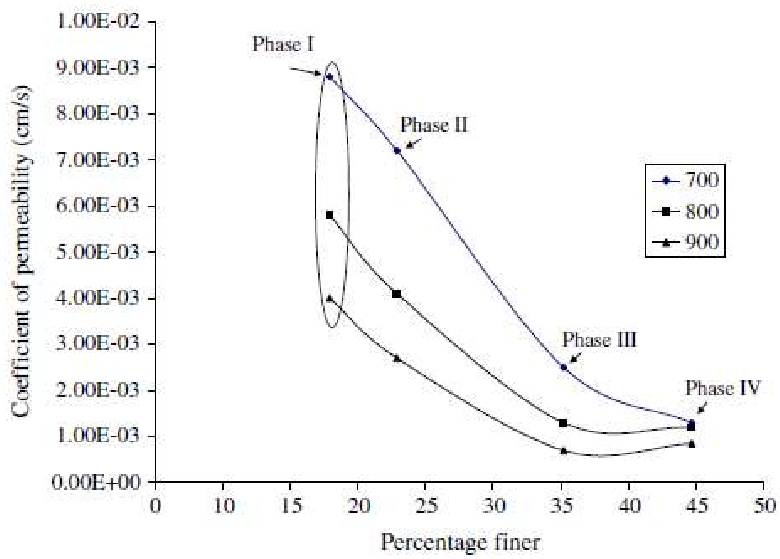


Figure 2-20 Permeability with the percentage of particles passing US Sieve No. 200 (Hossain et al. 2009)

2.4.4.7 Impact of landfill gas

Hudson et al. (2001) and Powrie et al. (2005) indicated a reduction of two orders of magnitude in hydraulic conductivity as a result of the in situ generation and accumulation of gas from tests. Powrie et al. (2008) investigated the effect of gas accumulation and pore water pressure on hydraulic conductivity at lower compression stresses, on sample of fresh shredded domestic waste (SW1) as shown in Figure 2-21. They performed tests at constant volume following initial compression under applied stresses of 40 kPa and 87 kPa, representing landfill depths of approximately 4 m and 9 m respectively. Figure 2–21 shows the changes in average bulk hydraulic conductivity and the volume of accumulated gas over a 27 day period for a test conducted at constant volume (corresponding to an initial applied stress of 40 kPa) with a relatively high average pore water pressure of 60 kPa. The reduction in hydraulic conductivity in response to the increase in the volume of gas contained within the sample is significant.

Powrie et *al.,* (2008) reported the results of hydraulic conductivity tests conducted at initial stresses of 40 kPa and 87 kPa under both high (60 kPa) and low (25 kPa) average pore pressure conditions. The results indicated that the hydraulic conductivity was reduced by gas accumulation, by approximately two orders of magnitude in low pore water pressure conditions and one order of magnitude in high pore water pressure conditions explained in Table 2-6

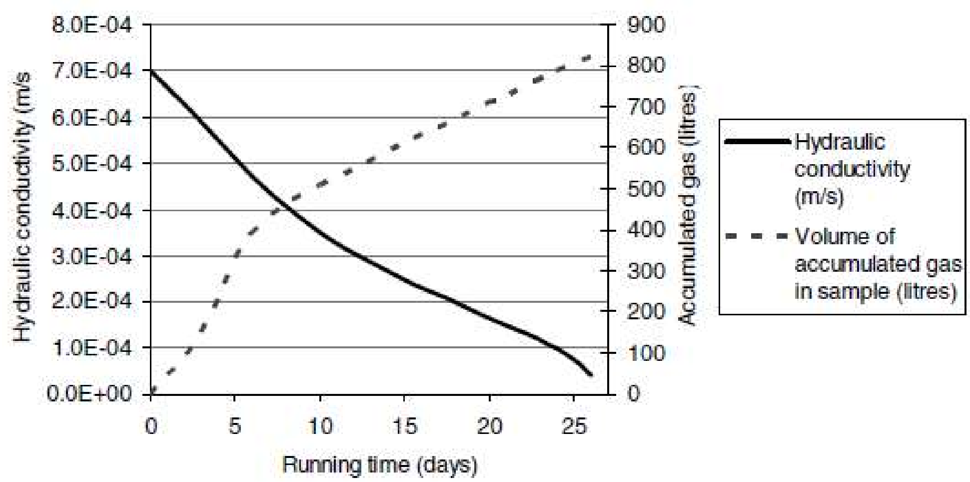


Figure 2-21 Changes in hydraulic conductivity with time for sample SW1 at a constant volume corresponding to an initial applied stress of 40kPa, with an average pore water pressure of 60 kPa (Powrie et al. 2008)

Table 2-6 Effect of gas accumulation on the hydraulic conductivity for sample SW1 (Powrie et al. 2008)

Applied stress (kPa)	Av. PWP (kPa)	Length of tests (days)	Accumulated volume of gas (liter)	Initial K (m/s)	Final K (m/s)	% of drainable pore volume occupied by gas (range)
40	25	36	1064	1.0×10^{-3}	1.5×10^{-5}	77.7−87.3
40	60	27	820	7.0×10^{-4}	4.0×10^{-5}	57.7−67.2
87	25	37	297	1.1×10^{-4}	1.1×10^{-6}	36.5−59.9
87	60	34	571	1.5×10^{-5}	1.2×10^{-5}	93.1−100

2.4.5 Porosity

There are not many studies available on the porosity published in the literature. This might reflect the difficulty of measuring porosity, especially in field conditions. The solid waste has different type of porosities such as total porosity, effective porosity and drainable porosity unlike soil. It has been mostly focused in drainable porosity of the waste in previous literature (Knox, 1992; Korfiatis and Demetracopoulos, 1984; oweis et al., 1990, Beaven, 2000). Bleiker et al. (1993) reported the porosity was not related to waste density, but was presented graphically as a relationship with hydraulic conductivity. Most of the previous studies explained the relationship between porosity and density of the waste

Liquid and gas transport in landfills are directly related to the structure of the porous medium (Oweis et al.1990; Beaven and Powrie, 1995; Bleiker et al., 1995; Durmusoglu et al., 2006; Jain et al., 2006). Thus, detailed characterization of MSW physical parameters is required prior to studying its hydraulic properties (i.e. liquid and gas permeability)(Stoltz et al., 2010). The density of the MSW is highly dependent on the compression stress which has been observed in numerous studies (Powrie and Beaven, 1999; Landva et al., 2000; Gource et at., 2001). In general, the hydraulic conductivity of any porous media is primarily a function of the interconnected pore space. In the case of soils, correlations have been observed between hydraulic conductivity and the void ratio. However, solid mass of MSW is a function of time and hence void ratio may not be the best parameter to explain the void space in MSW (Reddy et al., 2009b). Waste hydraulic conductivity and porosity influence the design and operation of bioreactor landfill. The heavily compacted waste has relatively lower permeability which hinders the leachate recirculation process in bioreactors (Khire and Mukherjee 2007). There are not many studies available about the relationship between hydraulic conductivity and porosity and between porosity and density. Since the bioreactor landfill is the addition of liquid in the MSW, it is necessary to understand the variation of porosity of the waste with the density. Liquid and gas transport through the porous medium so that porosity is one of the most important parameter for the distribution of liquid inside waste mass. Generally total and drainable porosity term were used for the waste by Beaven 2000, Husdon et al., 2001. Different types of porosities can be defined for the waste. Waste materials behave in a different way than soil. For example, if the soil is fully saturated, there will not flow occur in the downward direction due to gravity unless stress is applied but flow will occur under the effect of gravity in MSW if the sample is saturated without applying stress also. The stress application further increases the drainage. After certain time, equilibrium condition will occur and flow will stop. Depending upon this situation, porosities are defined for the MSW which are explained below.

2.4.5.1 Total porosity and void ratio

If the waste material is fully saturated all the void are assumed to be occupied with liquid. The ratio of volume of water to the total volume of waste is called total porosity of the waste. In order to estimate the total porosity, initially the samples are saturated and weight was recorded. The increase in weight will represent total water absorbed by the samples. Total porosity can be calculated using the equation (2.6).

$$n = \frac{V_v}{V_t} \text{.................................} \quad (2.6)$$

Where the volume (v_V) represents the void volume ($v_v = v_L + v_G$) and v_t represents the total volume of the sample (Solids and Voids).

Zeiss and Major (1993) determined total porosities in between 58% and 47% for waste compacted to densities between 170 kg/m^3 and 300 kg/m^3. It is not clear whether these densities are by wet or dry weight, but either way, they are rather small. As the densities are quite small, those might be dry densities of the waste. Zornberg *et al,*

(1999) reported a linear reduction in total porosity with depth in tests on landfill waste recovered from drill cores carried out in a 450 mm diameter compression cell.

Beaven (2000) reported a reduction in total porosity occurred with applied stress up to about 200 kPa, but negligible amount of reduction occurred with increasing stress after 200 kPa in a large 2 m diameter compression cell (Figure 2-22). The total porosity estimated approximately 45% at stresses above 200 kPa for the two wastes most representative of MSW in landfills (a fresh MSW and an aged MSW excavated from a landfill).

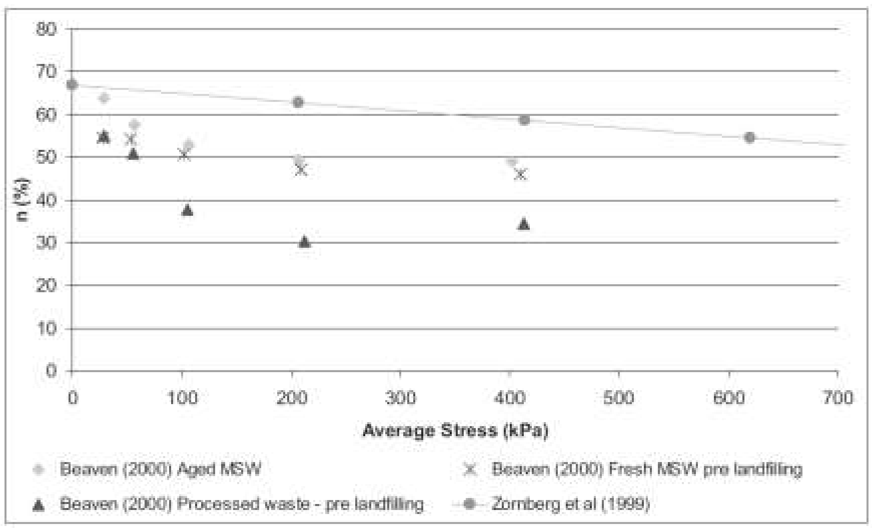


Figure 2-22 Relationship between total porosity and stress (Beaven et al. 2011)

Some authors (Gabr and Valero 1995, Stoltz et al 2010 b,) determined the void ratio of the waste in the laboratory. It is not available in the literature about the void ratio of the waste at various density and composition. It has been extensively used in the case of settlement and compression of the waste. Since it is directly related to the settlement, it becomes necessary to have a complete idea about the void ratio of waste at any condition. The total porosity can also be converted into void ratio. The relationship between void ratio and porosity can be expresses as:

$$e = \frac{n}{1-n} \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \text{(2.7)}$$

Where n is the total porosity of the MSW and e is void ratio

2.4.5.2 Retained/Effective porosity

If a fully saturated waste material is allowed to drain under gravity, its water content will decrease and ultimately reach a state which is called as field capacity. The ratio of volume of remaining water at field capacity to the total volume of waste is called effective or retained porosity. This porosity is basically filled with water all the times. In order to estimate the effective porosity, the saturated waste sample was allowed to drain for 24 hours under the gravity flow and the sample will attain a stable condition. The moisture content at this stage is also called the total absorptive capacity. The volume of pore space occupied with water per unit volume of waste sample is called effective porosity. Retained/Effective porosity can be calculated using the equation (2.8).

$$n_e = \frac{V_e}{V_t} \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \text{(2.8)}$$

Where v_e= volume of water at stable condition after gravity flow and v_t= total volume of waste

2.4.5.3 Drainable porosity

The drainable porosity is the same as the specific yield, which is well established in the hydrogeology literature. If a fully saturated waste material is allowed to drain under gravity, its water content will decrease as drainable pores empty. It will eventually reach a state (termed as the field capacity) when no further drainage occurs. The volume of freely draining water per unit volume of waste defines the drainable porosity. Theoretically the difference between total porosity and effective porosity gives drainable porosity. Drainable porosity can be calculated using the equation (2.9).

$$n_d = \frac{V_d}{V_t} \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \text{(2.9)}$$

Where, v_d= volume of drainage water and v_t= total volume of water

Most of the previous studies focused on the drainable porosity of the waste rather than total and effective porosity. Many authors did not consider the effective porosity of the MSW and treated drainable porosity and effective porosity as same porosity. It is necessary to clearly understand the definition of all porosities. Still there were not many studies available about drainable porosity in the waste. There are sparse data on drainable porosity published in the literature as compared to hydraulic conductivity of the MSW. This might reflects difficulty of measuring these parameters in field as well as in laboratory conditions.

Korfiatis and Demetracopoulos (1984) used refuse cylinders (0.56 m diameter by 1.8 m high) to investigate the unsaturated flow through the waste. Saturated water contents of between 50-60% (v/v), were determined for waste with a dry density of 440 kg/m^3. These data indicate that the waste had a drainable porosity of approximately 30%.

Beaven (1996) reported the results of a pumping test on a 9 m depth of refuse with a 5-6 m saturated zone which yielded a value of specific yield (s_y) of 4%. A further pumping test was performed on the same waste when landfilling depth increased to 23 m and the saturated zone had increased to 6-7 m. Surprisingly the calculated specific yield (drainable porosity) had increased to 7% in later test.

Burrows et al. (1997) reported the results from pumping tests of four landfill sites from UK. They performed more than 50 tests and the majority of the tests were of relatively short duration (3-8 hours) and drawdown data from pumped wells only were analyzed. However, there were also several tests of longer duration (between 2 and 4 weeks) on single wells and multiple pumped and observation well sets. Drainable porosities in the range 9-16% were determined, mainly from tests where data from observation wells were available.

Zornberg et al*.,* (1999) presented data on changes in volumetric water content with depth in a landfill, from which a relationship between drainable porosity values and stress may be derived. Their analysis indicated a linear reduction in drainable porosity occurred with depth (and stress) from approximately 15% at the surface to less than 3% at a depth of 60 m, which they equate to a stress of slightly over 800 kPa (Figure 2-23).

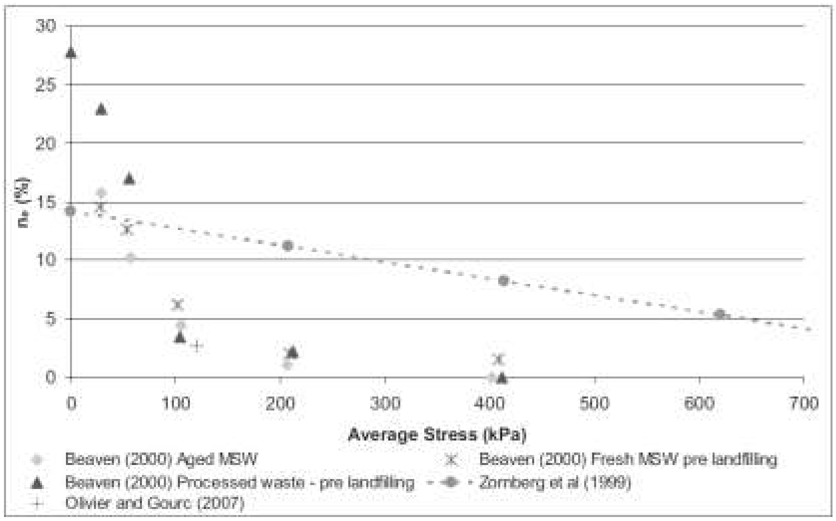


Figure 2-23 Relationship between drainable porosity and stress (Beaven et al. 2011)

2.4.6 Factors affecting porosity

The total porosity of MSW depends on density and stress, particle shape and size, waste composition, degradation level, and depth of burial within the landfill. Besides this, effective and drainable porosity depend on temperature, time of duration. Hence, along with the physical properties of the solid waste, the variation in porosities should be considered in order to design efficient leachate recirculation system. Basically it is one of the most important parameter along with hydraulic conductivity of the municipal solid waste. The porosity and hydraulic conductivity are well interconnected with each other in the case of soil. Similarly there might be a strong dependency between porosity and hydraulic conductivity of the waste. There were few studies available on the porosity of the solid waste. A detailed study is required in order to find out the relationship between porosity and other parameters of the waste such as density, stress, hydraulic conductivity at various compositions.

2.4.6.1 Stress and density

The relationships between porosity and the dry density of waste and stress level appear to be well defined. Many authors have presented data relating porosities to waste density and stress. Most of the studies were focused on the drainable porosity and the density /stress. Many authors (Beaven 2000; Zornberg et al., 1999; Oliver and Gource 2007, Staub et al., 2009) reported the change in porosity with the density or stress level which indicated that porosity decreased with the increased in density and stress (see Figure 2-22 and Figure 2-23).

2.4.6.2 Degradation

There were not any studies available in literature the relationship of porosity variation with degradation level. Since mass of the MSW will reduce with time due to degradation and settlement also will occur due to degradation in the same time make complex relationship between porosity and degradation level of the waste. But there is relationship between degradation level and particle size of the waste as was reported by Hossain et al., 2009. They performed particle size analysis of the solid waste at different stages of degradation. It had been found that MSW particles were relatively larger during the initial stages but with degradation, the matrix structure of paper; textile and other degradable constituents were broken down into smaller particles. This resulted in an overall increase in percentage of finer particles passing through US sieve 200 (0.074 mm). The percentage of finer particles passing through US sieve No. 200 (0.074mm) in ‘Phase I’ is only 10% and it increased to 39% in ‘Phase IV’.

2.4.7 Moisture holding capacity of MSW

In general fresh MSW can contain some water but cannot be always saturated for a long time. When the moisture content of the MSW exceeds its field capacity, which is defined as the maximum moisture that can be retained by waste mass without producing downward percolation, it start to drain that water (Beaven 2000). 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 concept of the bioreactor landfill is the addition of water in order to enhance the decomposition process. The understanding of moisture content distribution within municipal solid waste (MSW) in a landfill due to leachate recirculation plays an important role in designing and optimizing the operation of a leachate recirculation system. The increased moisture content is the major contributor to leachate formation because it is commonly associated with enhancing biodegradation processes in landfills.

Furthermore, as the landfilling concept moving from a conventional (dry cell) concept to a bioreactor (wet cell) approach, there is a growing need for in situ monitoring of landfills to maintain waste at moisture content favorable to maximize the biodegradation and gas generation.

Moisture information is important in hydrological studies to detect changes of moisture storage in water balance analysis, assess the performance of leachate recirculation systems, and investigate moisture flows in MSW. For a landfill operator, it would be an advantage 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).

In soil mechanics the water content is defined as the ratio of mass of water to the mass of dry solid. It is normally denoted by "w" for soil in soil mechanics. However, two types of moisture content were used for the solid waste.

On dry basis: The water content is defined as the ratio of mass of water to the mass of dry solid and is denoted by wc_{dry}.

$$wc_{dry} = \frac{M_w}{M_s} \qquad (2.10)$$

On wet basis: The water content is defined as the ratio of mass of water to the total mass of water and solids.

$$wc_{dry} = \frac{M_w}{M_s + M_w} \qquad (2.11)$$

The relationship between the two water content can be expressed as,

$$wc_{dry} = \frac{wc_{wet}}{1 - wc_{wet}} \qquad (2.12)$$

$$wc_{wet} = \frac{wc_{dry}}{1 - wc_{dry}} \qquad (2.13)$$

The water content of the MSW can also be expressed on a volumetric basis. The volumetric water content "wc_{vol}" is defined as the ratio of the volume of water to the total volume of air, solids and water.

$$wc_{vol} = \frac{V_w}{V_t} = \frac{M_w}{V_t \times \rho_w} \qquad (2.14)$$

2.5 Biodegradation of MSW in landfills

In this section, background on landfill gas generation, stages and phases involved during the biodegradation process, generation of leachate and gas, factors affecting on degradation process, enhancement methods of degradation in anaerobic bioreactor landfills, and case studies are discussed. Numerous studies on the anaerobic biodegradation process (e.g.: Barlaz et al.,1989; Warith, 2003; Warith et al., 2005; Christensen et al.,1989; Chiemchaisri et al., 2002; and Reinhart and Al-Yousfi,1996) are reported in the literature.

2.5.1 *Landfills gas generation*

Municipal solid waste (MSW) disposed of in a landfill is comprised of several types of waste constituents such as food, paper, wood/yard, plastic, textiles and leather, Styrofoam , C& D and metal waste. Out of those waste components, many waste components consist of high organic content which can decompose through the microbial activity. The organic fraction of municipal solid waste in the landfill decomposes through a series of interacting microbial processes and produces the methane (CH_4), carbon dioxide (CO_2) water (H_2O) and several other traces materials. Methane and carbon dioxide are the main byproducts of the waste decomposition. Landfill gas is a complex mixture of different gases produced by the action of microorganisms within a landfill.

The rate of methane production from landfills 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 also recirculated water/leachate content, as well as climatic factors such as the annual rainfall and temperature.

The heterogeneity of the MSW, together with the frequently unclear nature of the contents, makes landfill gas production more difficult to predict. Landfill gas consists of approximately forty to sixty percent methane, with the remainder being mostly carbon dioxide. The content of methane gas is so high that it is potentially an energy source as well as a greenhouse gas. According to IPCC (2004), methane has 22 times more global warming potential than carbon dioxide (over a hundred year time period). Landfill gas also contains varying amounts of nitrogen, oxygen, water vapor, hydrogen sulphide, and other contaminants. Typical composition of landfill gas is shown in Table 2-7. Most of these other contaminants are known as "non-methane organic compounds" or NMOCs. Municipal solid waste usually produces the non-methane organic compounds less than one percent of landfill gas.

The Environmental Protection Agency estimates that there are approximately six thousand landfills in the United States. Most of these landfills are composed of municipal waste, and, therefore, produce methane. These landfills are the largest source of anthropogenic methane emissions in the United States. In addition, USEPA (2008) has identified above 100 trace constituents including non-methane organic compounds (NMOCs) and volatile organic compounds (VOCs) emitted from landfills. USEPA (2005) User's Guide for Landfill Gas Emissions model incorporates default emission factors for 46 trace components.

Table 2-7 Typical composition of landfill gas (Tchobanoglous et al. 1993)

Components	Percent,% (Dry volume basis)
Methane, CH_4	45-60
Carbon dioxide, CO_2	40-60
Nitrogen, N_2	2-5
Oxygen, O_2	0.1-1
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

2.5.2 Biodegradation stages

Shortly after MSW is landfilled, the organic components of MSW start to undergo biochemical reactions. In the presence of atmospheric air, which is near to the surface of the landfill, the organic compounds are oxidized aerobically. Landfill gas is a product of the natural biological decomposition of organic material contained in wastes deposited in landfills. The latter includes paper, animal and vegetable matter, garden wastes, and food, wood, plastic, textiles and leather, Styrofoam, C& D and metal wastes. The production of the principal landfill gas components occurs in four more or less sequential phases, with the final phase being characterized by the constant production of methane

(60%) and carbon dioxide (40%) (Farquhar and Rovers, 1973). The latter gases, which are assigned the generic description of “landfill gas”, continue to be produced until the majority of the organic material in the waste has been degraded.

During this process, the organic content of MSW is decomposed and converted into biogas basically in two stages: (i) aerobic stage and (ii) anaerobic stage. Certain types of bacteria groups are developed in each stage and breaking down reaction will take accordingly.

2.5.2.1 Aerobic stage

Soon after waste disposal, the organic components start to undergo biochemical reaction for a short period of time due to primarily the presence of oxygen within the waste voids. The duration of reaction primarily depends on the amount of oxygen that is available inside the MSW. The amount of oxygen in the waste basically depends on compaction effort in MSW disposal. Loosely compaction leads to more porous in waste and thus can store greater amount of oxygen than in high compaction. In this stage, organic components are oxidized in the presence of aerobic bacteria to produce carbon dioxide and water vapor (Themelis and Ulloa, 2007). After this stage, biochemical reaction of the waste will shift to the anaerobic stage due to the depletion of oxygen.

2.5.2.2 Anaerobic Stage

The conversion of waste to methane and carbon dioxide is aided by microorganisms by a series of chemical conversion processes. The principle bioreaction in landfills is anaerobic digestion that takes place in three stages. In the first, fermentative bacteria hydrolyze the complex organic matter into soluble molecules. In the second, these molecules are converted by acid forming bacteria to simple organic acids, carbon dioxide and hydrogen; the principle acids produced are acetic acid, propionic acid, butyric acid and ethanol. Finally, in the third stage, methane is formed by methanogenic bacteria, either by breaking down the acids to methane and carbon dioxide, or by reducing carbon dioxide with hydrogen (Barlaz et al., 1990; Christnensen and Kjeldesn 1989; Themelis and Ulloa 2007).The overall process of converting organic matter to methane and carbon-dioxide can be expressed as shown in two representative reactions.

Acetogenesis

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

Methanogenesis

$$CH_3COOH \rightarrow CH_4 + CO$$

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

2.5.3 Phases of MSW degradation in landfill

Numerous studies have been carried out on the anaerobic biodegradation process in the landfills. Numerous researchers (Christensen and Kjeldsen, 1989; Barlaz, et al., 1989a) 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 MSW biodegradation process have been reviewed by many researchers 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). Some studies suggested that the biodegradation of MSW can be divided into five distinct phases (Warith et *al.,* 2005). Landfill gas composition and leachate concentration vary from one phase to another. The rate and characteristics of produced leachate and biogas vary from one phase to another, and reflect the microbially mediated processes taking place inside the landfill waste (Reinhart and Al-Yousfi, 1996). The phases experienced in the process of degradation are described below (Warith et *al.,* 2005).

Phase I: Initial adjustment phase (Aerobic phase)

This phase will start immediately after disposing the waste in landfill. In this phase, both oxygen and nitrate are consumed, with soluble sugars serving as the carbon source for microbial activity. The quantity of oxygen available is low which is basically depends on compact level. In addition, this phase is associated with initial placement of solid waste and accumulation of moisture within landfills. An acclimation period (or initial lag time) is observed until sufficient moisture develops and supports an active microbial community (Reinhart and Al-Yousfi, 1996).

Phase II: Transition phase

In this phase, a transformation from an aerobic to anaerobic condition occurs with the depletion of oxygen trapped within a landfill, and the anaerobic microorganisms become active. The hydrolytic and fermentative microorganisms hydrolyze polymers such as carbohydrates, fats, and proteins. The initial products of polymer hydrolysis are soluble sugars, amino acids, long-chain carboxylic acids, and glycerol (Barlaz et al. 1990). By the end of this phase, measurable concentrations of COD and volatile organic acids can be detected in the leachate (Reinhart and Townsend 1998).

Phase III: Acid formation phase

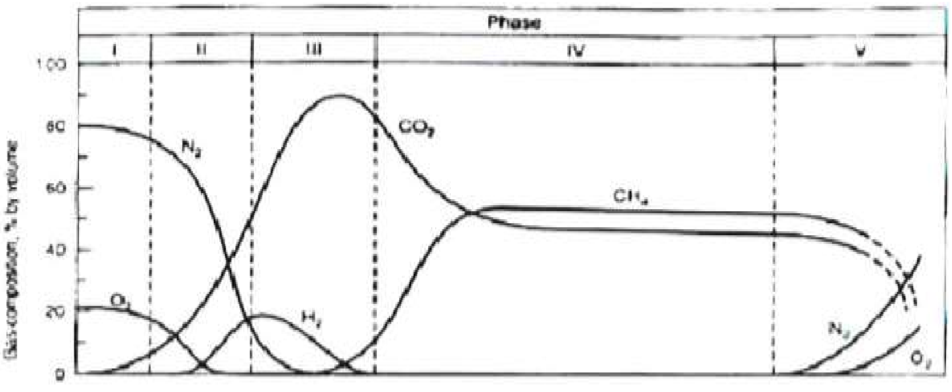
During the first stage of this phase, the intermediates produced from Phase II are further fermented into short-chain carboxylic acids, carbon dioxide, and hydrogen. Acetate and alcohols are also formed. During the second stage of this phase, the obligate proton-reducing acetogens become active. They oxidize the fermentation products of the first stage to acetate, carbon dioxide, and hydrogen. The conversion of short-chain carboxylic acids to acetate is only thermodynamically favorable at very low hydrogen concentration. However, there is a hydrogen scavenging population, i.e., methanogens in an active anaerobic ecosystem. If fermentative and methanogenic activities are not balanced, intermediates will accumulate and may percolate from the landfill as leachate (Barlaz et al. 1990).

Phase IV: Methane fermentation phase

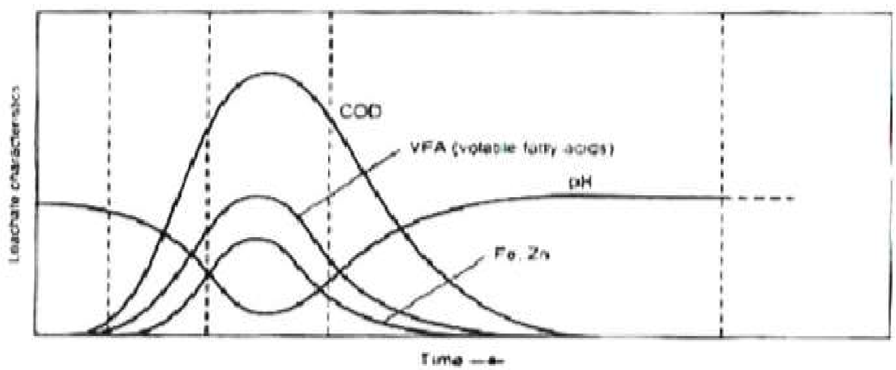
During this phase, both methanogens and sulphate reducing bacteria are become active in the degradation process. The hydrophilic methanogenic bacteria transform hydrogen and carbon dioxide into methane, while the acetophilic methanogenic bacteria transform acetic acid into methane and carbon dioxide. The rate of methane gas production increases rapidly to some maximum value. Methane gas constitutes approximately 50-60% (by volume) of gas composition (Barlaz, et al., 1990; Warith and Sharma, 1998). The pH is increased, and consequently heavy metals are removed by precipitation. The organic contents present in the leachate declines.

Phase V: Maturation phase

During this phase, the biodegradable organic matter is stabilized, and nutrients become limiting. Gas production drops dramatically and leachate strength stays steady at much lower concentrations. Concurrently, there is an increase in the rate of cellulose plus hemicellulose hydrolysis. MSW degradation can take 30 to 100 years in traditional landfill however, with leachate recirculation, the whole process is accelerated with degradable waste or higher gas production/recovery potential and more stable leachate during subsequent methane fermentation phase is encountered at bioreactor landfill. Typical phases in waste degradation are shown in Figure 2-24 (a, b).



(a)



(b)

Figure 2-24 Phases of degradation in a typical landfill (a) for gas (b) for leachate (WMI 2000)

2.5.4 Landfill leachate

Landfill leachate is generated by excess rainwater percolating through the waste layers in a landfill. A combination of physical, chemical, and microbial processes in the waste transfer pollutants from the waste material to the percolating water (Christensen and Kjeldsen, 1989; Kjeldsen et al. 2002). The major potential environmental impacts related to landfill leachate are pollution of groundwater and surface water. The risk of groundwater the pollution is probably the most severe environmental impact from landfills

2.5.4.1 Leachate generation in landfills

Leachate is produced when the moisture content of the waste exceeded its field capacity. The generation of leachate is primarily caused by precipitation percolating through waste deposited in a landfill. The percolating moisture becomes contaminated when it comes in contact with waste and if it then flows out of the waste material it is termed as leachate. Additional leachate is also produced during the biodegradation process of solid waste including others products as methane, carbon dioxide, organic acids. Leachate generation in the landfill is the result of precipitation, evaporation, surface runoff, infiltration, storage capacity. Under humid climatic conditions the average difference between precipitation and evaporation also with different vegetation covers is positive. Beside evaporation the infiltration could be reduced by surface runoff. But preventing of erosion problems needs a limitation of surface runoff. Several studies are available about the leachate generation from landfills (Reinhart, 1996; Rees, 1980; Kjeldsen et al., 2002; and El-Fadel et al., 1997a). The Leachate generation is affected by several factors, including initial moisture content of waste, amount of water recirculating the waste, climatic condition, composition and type of waste, and density of waste (El-Fadel el al., 1997a; and Rees, 1980).

2.5.4.2 pH of leachate

Leachate is a liquid that has percolated through solid waste and includes extracted, dissolved, and suspended materials that may include potentially harmful materials. The type of solid waste, physical, chemical, and biological activities that occur in the landfill determines the quality of leachate. The acidogenic bacteria has a wider pH range as compared to the methanogenic bacteria. The methanogenic bacteria produce methane and carbon dioxide gas from hydrogen gas and acid in MSW, and this process occurs in environmental conditions 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 would be slowed down. According to Christensen et al. (1996), if the methanogenic activity is limited, conversion of acetic acid to methane and carbon dioxide decreases and acids accumulate. This leads the pH value to decrease and this may stop or slow down the methane production.

2.5.4.3 Leachate composition

Landfill leachate consists of many different organic and inorganic compounds that may be suspended or dissolved. Leachate composition depends on various factors including waste composition, age of the waste, phase of decomposition, temperature and land filling technology. The type of municipal solid waste, physical, chemical, and biological activities that occur in the landfill determines the quality of leachate. During the decomposition leachate is generated by excess rainwater infiltrating the waste. The leachate contains four groups of pollutants: dissolved organic matter, inorganic macrocomponents, heavy metals, and xenobiotic organic compounds (Kjeldsen et al. 2002). Kjeldsen et al. (2002) performed a detailed study about the composition of leachate. According to authors 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 for example cadmium (Cd2+), chromium (Cr3+), copper (Cu2+), lead (Pb2+), nickel (Ni2+), and zinc (Zn2+). The variation of leachate composition with the biodegradation phases are depicted in Table 2-8.

Table 2-8 Leachate composition for different phases (Kjeldsen et al., 2002)

Parameter	Acid 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		
Sulphate	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	

2.5.4.4 BOD and COD

The BOD to COD ratio is also an indicator of the proportion of biologically degradable organic matter to total organic matter. This ratio decreases with the age of 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 1750 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 variation on the reactors. According to the author, the overall, 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 reported the BOD concentration increased in all bioreactors at the beginning of the anaerobic stage, as a result of low methanogenic activity which facilitated the accumulation of organic acids from the hydrolysis and acidogenesis steps. The author observed the BOD peak reduction showed the following sludge addition enhanced the biodegradation of MSW.

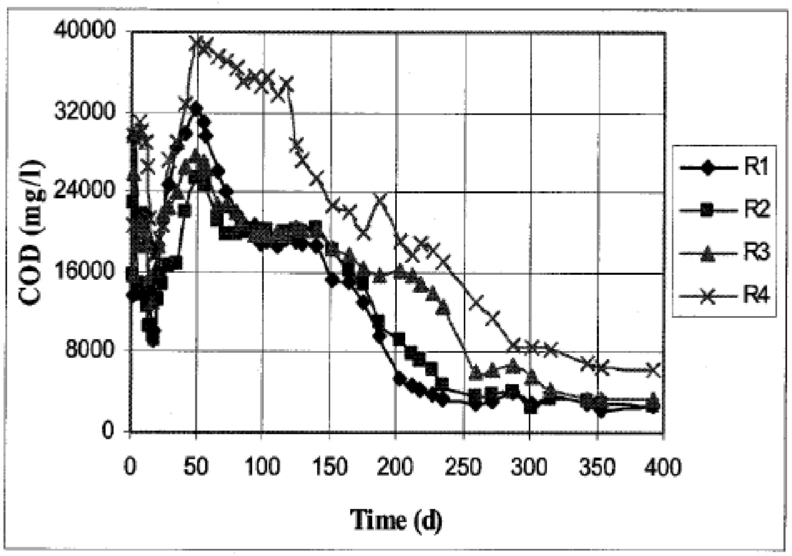


Figure 2-25 COD variation (al-kaabi 2007)

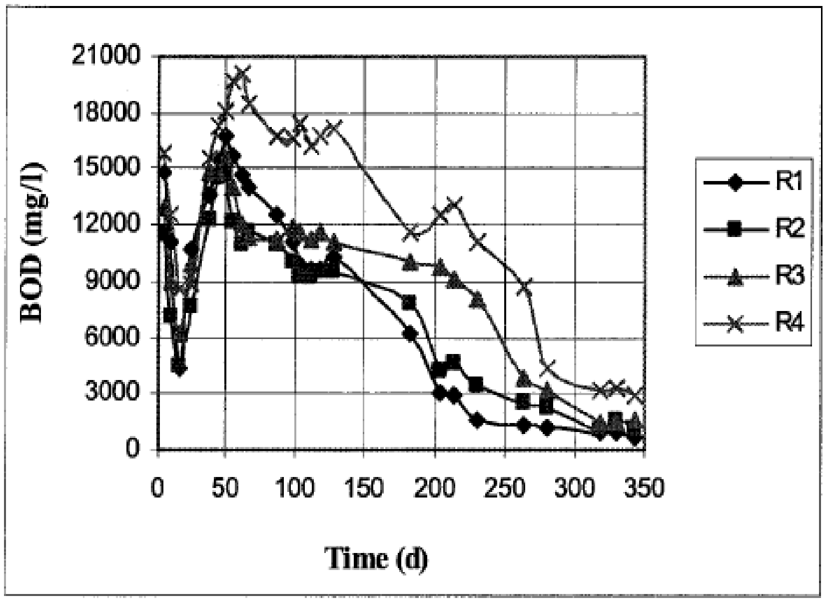


Figure 2-26 BOD variation (al-kaabi, 2007)

2.5.5 Factors affecting the biodegradation of MSW

Several studies showed the main agent to accelerate the biodegradation of waste are the addition of water and leachate. Among the technologies applied in bioreactor landfill for enhancing waste biodegradation, moisture control through leachate recirculation has been proven to be the most effective and practical strategy (Warith et al., 2005). Numerous practices have already showed that the leachate recirculation in landfills can potentially lead to more rapid waste decomposition, stabilization and settlement. The factors controlling biodegradation are water content, pH, nutrients, oxygen concentration, hydrogen concentration, temperature, inhibitors, leachate recirculation, sludge addition, and waste composition (Barlaz et al., 1982; Barlaz et al., 1989, Christensen et al., 1992; El-Fadel et al., 1996; Wraith, 2003; and Wraith et al., 2005).

2.5.5.1 Leachate recirculation and moisture control

Leachate recirculation appears to be the most effective technique to increase moisture content inside landfill in a controlled fashion. The advantages of leachate recirculation include distribution of nutrients and enzymes, pH buffering, dilution of inhibitory compounds, recycling and distribution of methanogens, liquid storage and evaporation opportunities (Reinhart et al 1996). It has been suggested that leachate recirculation could reduce the time required for landfill stabilization from several decades to 2-3 years, thus minimizing the opportunity for long-term adverse environmental impact (Pohland 1975).

Moisture content is considered the most important factor in solid waste decomposition and gas production. It provides the aqueous environment necessary for gas production and also serves as a medium for transporting nutrients and bacteria throughout the landfill. If the moisture content in the waste exceeds the field capacity, the moving liquid will carry nutrients and bacteria to other areas within the landfill, creating an environment favorable to increase gas production. A number of studies have confirmed that methane generation rate increases with an increase in waste moisture content (Barlaz et *al.,* 1990; Mehta et *al.,* 2002; Wreford et *al.,* 2000; Alvarez and Martinez-Viturtia, 1986; Chan et *al.,* 2002; Lay et *al.,* 1998). Water is the key factor to accelerate the biochemical decomposition of organic substances (Pohland, 1970; Lechie and Pacey, 1979; Klink and Ham, 1982). Moisture content is the single most important factor that promotes the accelerated decomposition. The bioreactor technology relies on maintaining optimal moisture content near field capacity (approximately 35 to 65%) and adds liquids when it is necessary to maintain that percentage (US EPA). Moisture content in the waste is a crucial factor for the microbial activity and methane gas production. The effect of increased moisture content limits the oxygen transport from the atmosphere; facilitates exchange of substrate, nutrients, buffer, and dilution of inhibitors; and spreads micro-organisms inside the landfill (Christensen et al., 1996a; Warith et al., 2005). Rees (1980) summarized that by increasing water content from 25% to 60% the gas production rate and the percentage of methane are increased. In bioreactor landfills, the biodegradation of MSW is accelerated via active leachate recirculation and/or liquid addition, which increases moisture content and transports nutrients to microorganisms (Pohland and Kim 2000; Mehta et al. 2002; Kim and Pohland 2003; Soong et al. 2009). Degradation of MSW leads to landfill gas generation, a sustainable energy source, waste volume reduction, and settlement. Moisture content is the most important factor controlling the degree and rate of MSW biodegradation (Reinhart and Townsend 1998; EPA 2006; Barlaz et al. 2010a). Reinhart and Al-Yousfi (1996) reported that the leachate recycle not only improves the leachate quality, but also shortens the time required for stabilization from several decades to 2–3 years.

The moisture movement through a decomposing solid waste sample appears to increase methane gas generation rates by 25 to 50% over methane gas rates observed during minimal moisture movement but at the same overall moisture content levels. This point out the difference between moisture content and moisture movement as two separate variables affecting methane generation rates (Klink 1982). Faruquhar and Rovers (1973) reported a critical review of the factors affecting methane generation in landfills, and found that maximum methane was generated at moisture contents of 60% to 80% on wet weight basis.

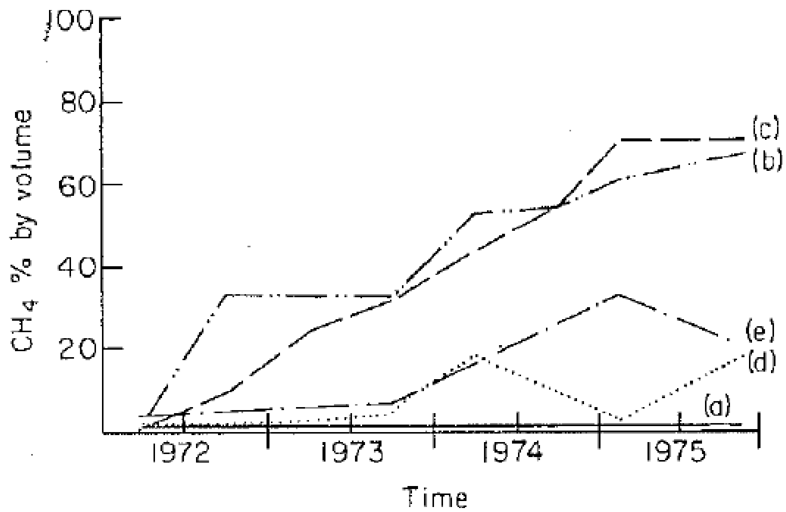
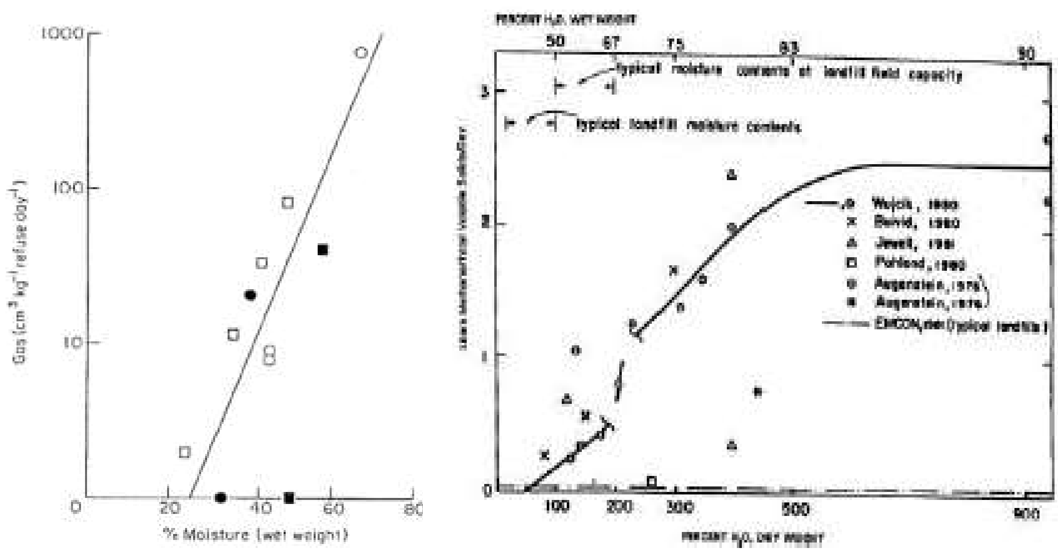


Figure 2-27 Effects of water content on the methane content of landfill gas (a) Dry waste; (b), (c) Daily liquid application; (d),(e) Initially saturated (Rees, 1980)



(a) (b)

Figure 2-28 Plots of moisture content vs. methane generation rate by (a) Rees (1980) and (b) SWANA (1998)

Rees (1980) observed effect of moisture circulation on wastes and reported daily circulation of water yielded maximum methane content which is shown in Figure 2-27. The author reported daily recirculation of leachate/water increased the methane content on gas generation. Rees (1980) plotted the methane generation and moisture content data published in research papers and found that the log of methane generation rate produced from landfills was directly proportional to the moisture content (Figure 2-28 a). Similarly, Solid Waste Association of North America (SWANA, 1998) also developed a graph of varying rate of methane generation with moisture content of waste which showed a linear relationship between log of rate of methane generation rate and moisture content of waste until the waste reaches saturation state. However methane generation rate was assumed to be constant, irrespective of the moisture content of waste (See Figure 2-28 b).

Hernandez-Berriel et al., (2010) studied the effect of various moisture contents on the methane generation rate and leachate characteristics on laboratory scale bioreactors operated with leachate recycling process. Four moisture contents 50%, 60%, 70% and 80% were considered to observe the effect. It was found that the methane generation rate increased as the moisture content increased from 50% to 70%, with 70% moisture content producing the maximum amount of methane. However, the methane generation rate was dropped in the 80% moisture content, presumably due to washout of nutrients.

2.5.5.2 Density/Stress

There is very limited research going on about the effect of density and stress on the bio-degradation of the solid waste. Most of the current research focused on the settlement of waste due to degradation. The density of the waste is an important parameter which affects many properties of the solid waste such as hydraulic conductivity, porosity and strength parameters. As the density is such an important factor it might affect bio-degradation of waste. Most of the gas generation models did not consider the density factor in the gas generation. Since the bioreactor landfill is operated with the addition of moisture, it should be design based on the moisture flow in order to maintain the minimum moisture content. For this reason, density should be a most important criterion while designing the bioreactor landfill.

Liu et al., 2010 studied the effect of stress on the biodegradation and settlement behavior of municipal solid waste (MSW). Three biodegradation-compression test apparatuses were developed for different stress level. The apparatuses were equipped with units for vertical loading, leachate recirculating, gas collecting, and temperature controlling. Fresh MSW samples were compacted the prescribed density in the test apparatus, and then subjected to three different stress levels to 100, 200 and 400 kPa. Biodegradation processes of the MSW samples were simulated at the controlled temperature (41°C) and under a leachate recirculation condition. The experimental results indicated that the gas production rate of the MSW was independent to the stress level (Figure 2-29) whereas the secondary compressibility parameters of the MSW were dependent on the vertical stress level (Figure 2-30)

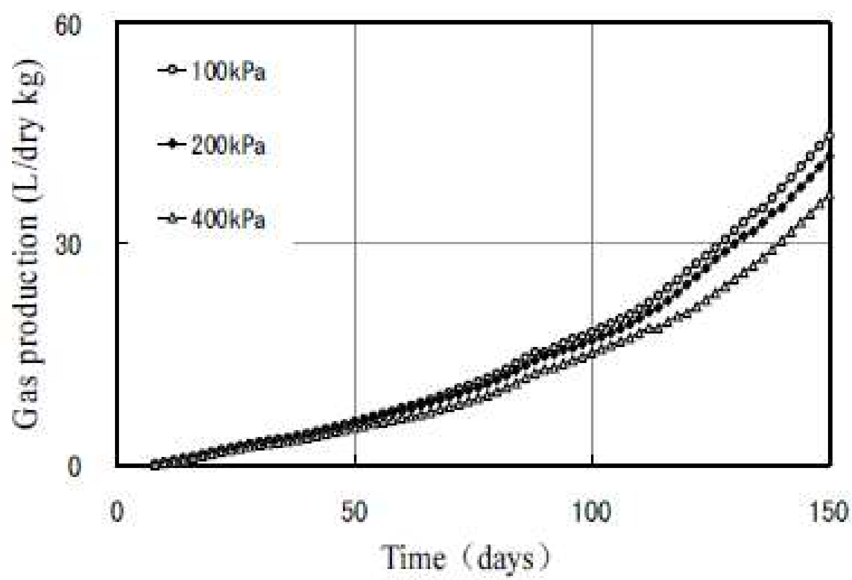


Figure 2-29 Cumulative biogas production of the samples (Liu et al. 2010)

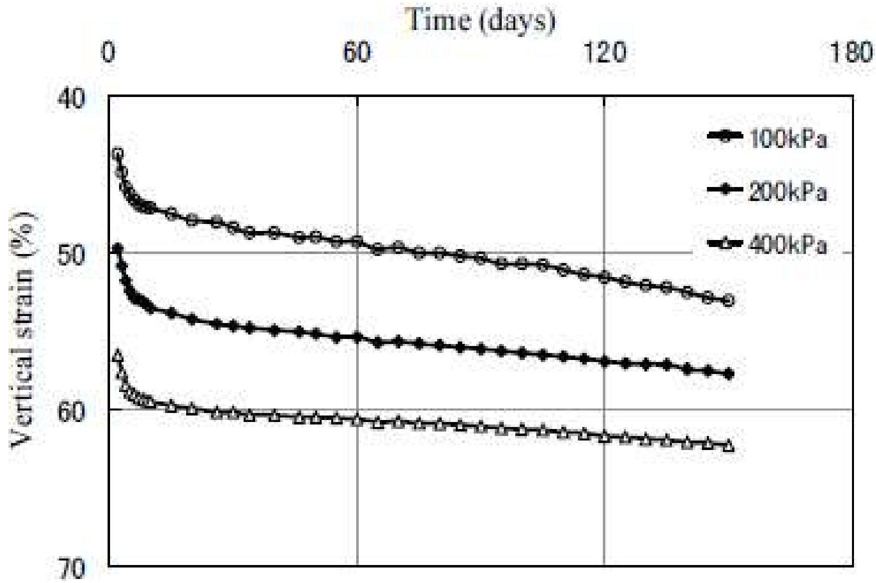


Figure 2-30 Variation of settlement with time (Liu et al. 2010)

2.5.5.3 Composition

Waste composition changes with geographical location, depending on economic conditions, lifestyle of people, industrial structure and waste management techniques. The amount of methane generated from a landfill depends on the organic content of the waste. The waste consists of several components which has varying amount of organic contents. Further, different components of waste degrade at varying rates. Hence, the rate at which methane is generated from landfills also depends on the waste composition.

Several studies are available to estimate the gas generation from the landfills. Various models have been developed to describe landfill waste degradation, including zero-order, first-order, second-order decay models, multiphase models, and combination models. Two of the most commonly used first-order and multi-phase first-order models are the U.S. Environmental Protection Agency's (EPA's) Landfill Gas Generation Model (LandGEM) and the Intergovernmental Panel on Climate Change (IPCC's) methane generation models, respectively (U.S. EPA 2005; IPCC 2006). IPCC guidelines (2006) recommended the use of a "multiphase first-order decay model" for estimation of methane emissions from landfills (Eggleston et al. 2006). A simplified version of the multiphase model is shown in equation 2.15. The landfilled waste is divided into categories: slowly-degrading, moderately-degrading, and rapidly-degrading waste respectively.

$$Q_{CH_4} = M_i \times L_0\left[F_r\left(k_r \times e^{-k_r(t-t_i)}\right) + F_m\left(k_m \times e^{-k_m(t-t_i)}\right) + F_s\left(k_s \times e^{-k_s(t-t_i)}\right)\right] \text{.......... (2-15)}$$

Where,

Q_{CH4}= methane emission rate, m^3/yr

L_0 = methane generation potential, m3 of CH_4/ Mg refuse

Mi = mass of waste in i^{th} section (annual increment), Mg

F_r, F_m , F_s = fraction of rapidly, moderately or slowly decomposing wastes

k_r, k_m, k_s= first-order decay constants for rapidly-, moderately- or slowly-decomposing waste

t_i = age of i^{th} increment in years

The variables k_r, k_m and k_s are assumed to be dependent on waste composition and other environmental factors such as moisture content, ambient temperature, and the depth of the landfill, while L_0 is assumed to be dependent of the waste composition. IPCC's methane generation model is based on the amount of degradable organic matter (DOC_m) in the waste disposed. The amount of degradable organic matter (DOC_m) in the waste is estimated from the information about the waste deposited in the landfill, and its components such as paper, food waste, yard waste, and textile. The decomposable degradable organic matter (*DDOC*) is defined as the amount of DOC that can be degraded in a landfill under anaerobic conditions and can be calculated as shown in equation 2.8.

$$DDOC_m = W \times DOC \times DOC_f \times MCF \ldots\ldots\ldots\ldots\ldots\ldots (2.16)$$

Where,

$DDOC_m$ = mass of decomposable DOC deposited (Mg)

W = mass of waste deposited (Mg)

DOC = degradable organic carbon in the year of deposition (Mg C/ Mg waste)

DOC_f = fraction of DOC that can decompose under anaerobic conditions;

MCF = methane correction factor for aerobic decomposition (before anaerobic decomposition starts) in the year of deposit

Karanjekar (2012) performed research on methane generation rate considering variable composition, temperature and rainfall. The author developed a Capturing Landfill Emissions for Energy Needs (CLEEN) model by incorporating the comprehensive regression equation into first-order decay based model for estimating methane generation rates from landfills. The author included the methane recovery and methane oxidation factors in the CLEEN model, to estimate the methane emissions from the landfill surfaces. A scale-up factor was computed to adapt the lab based regression equation to actual landfill scale methane generation using the City of Denton's landfill emissions data, which was found to be 0.012. The CLEEN model will allow better estimation of the methane generation rate constant (*k*) based on waste composition, rainfall and ambient temperature.

2.5.5.4 Temperature

The landfill temperature, moisture content/additive water amount, water characteristics (i.e., precipitation rainfall or other water entering landfills), available oxygen and waste characteristics are among the many factors contributing to gas and leachate generation at landfill sites and subsequently determine the characteristics of LFG and landfill leachate (El-Fadel et al. 1997). Yesiller et al. (2003) studied the spatial distribution of temperature over time in a landfill located in Michigan, US. They concluded that the temperature of waste is significantly affected by seasonal variations, placement of waste, age of waste and depth and location of waste together with moisture content of waste.

Most of the microbial activities are affected by temperature. There are always some temperature ranges which increase microbial activity and thus increase the methane generation rates. Anaerobic degradation is considered to be an exothermic reaction, although the heat generated during anaerobic degradation is only 7% of that generated during aerobic degradation. Hence the temperature in a landfill is expected to be higher than the atmospheric temperature (Christensen and Kjeldsen 1989; Rees 1980; Bingemer and Crutzen 1987).

Temperature is one of the most important parameter affecting the biodegradation, gas emission and gas generation. The landfill temperature is affected by the size and height of the landfill, climatic conditions and landfilling operations, which determine the circumstances in which microbial decomposition occurs (Wang et al. 2012). Understanding the impact of temperature on landfill gas emissions, especially landfill leachate, is significant for the improvement of long-term landfill management techniques, in order to minimize gas emissions, accelerate waste stabilization and shorten the post closure time. The leachate quality varies significantly in the transition from acidogenesis to methanogenesis. The biodegradation rate increases with temperature, but within certain limits.

Rees (1980) identified that the necessity of maintaining temperatures of about 45°C in a conventional anaerobic landfill. Similarly, Hartz et al. (1982) investigated the seven different temperatures ranged from 21°C to 48° C and observed that 41° C was the optimum temperature for short-term methane production. Mata-Alvares and Martina-

Verdure (1986) reported that the optimum temperature was 34° C to 38° C for MSW degradation but was independent of leachate recirculation. Blakey et al. (1997) reported that temperature is an important factor affecting the methane content of landfill gas. The operation of landfills under optimum temperatures will result in increase in the rate of gas production and refuse stabilization. Besides this, the transition from the acetogenic to methanogenic phase can be shortened when the landfill is operated in warmer climates. Robinson (2007) summarized that the transition period from the acetogenic to the methanogenic phase of the landfills in temperate countries was two or three times that of landfills in warmer climates. High methane production and a rapid transition from acidogenesis to methanogenesis can reduce the content of VFAs (volatile fatty acid) in leachate, rendering low BOD and BOD/COD ratios.

Christensen and Kjeldsen (1989) observed that the methane production rate increases when the temperature is raised from 20 to 30 and 40°C in laboratory scale tests. The higher temperature might not be friendly for microbial activity in the waste. Significant reduction in methane generation was observed with temperature less than 20°C and greater than 70°C (Tchobanoglous et al. 1993). Buivid et al. (1981) also studied the effect of temperature on waste degradation in laboratory scale landfill reactors. Three temperatures were chosen, 25°C, 37°C and 60°C. The authors reported that 37°C was the most favorable temperature for enhanced methane generation.

It can be concluded that it is extremely difficult to guess the temperature within a landfill, which is affected by a number of factors such as waste age, depth, proximity to the landfill's edges, temperature during placement, and moisture content. However, the temperature acts as both a stimulator and response to biodegradation. The higher temperature measurements observed in the landfills may be due to the presence of anaerobic microorganisms. It is, however, crucial to study the effect of temperature increase on the rate of biodegradation, and whether temperature together with moisture affects the rate of biodegradation.

2.5.5.5 pH Level

The pH of the recirculated leachate significantly influences the chemical and biological processes of the waste. Recent studies had shown that methanogenesis was favorable over a pH range 6.4–7.2 (Chugh et al., 1998; Yuen et al., 2001). Valencia et al. (2009) reported that pH was the possible ‘driving force’ to trigger all processes. The ideal methanogenic bacteria activity occurs in environmental conditions within a pH range of 6.8 to 8.0 (Warith, 2003). Any drop in the pH value below 6.8 will slow down the activity and growth of methanogenic microorganisms. In a well-established methanogenic media, if the methanogenic activity is inhibited by other factors [O_2, H_2, etc.], the conversion of acetic acid to CH_4 and CO_2 decreases and leads to an accumulation of the acids, thereby decreasing the pH which in turn may stop the generation of methane (Christensen et al., 1996). Within the optimum pH range, methanogens grow at high rate leading to 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). During the initial stages of anaerobic decomposition, organic acids formation occurs and results in an acidic pH. As these organics begin originate, the pH should rise as the acids are converted to methane.

2.5.5.6 Nutrient content

Microorganisms in the landfill require various nutrients for their activity, such as carbon, hydrogen, oxygen, nitrogen, phosphorus, sodium, potassium, calcium, magnesium and other trace materials. Rees (1980) and Christensen et al.,(1996) found from existing literature that all the necessary nutrients and traces of heavy metals are available in most landfills, but heterogeneous insufficient mixing of the wastes may result in nutrient limited environments. These nutrients are found in most landfills. However, inadequate homogenization of the waste may result in a nutrient limited environment. Toxic materials such as heavy metals can slow the bacterial growth and consequently retard gas production. If there is the greater the amount of digested nutrients, the greater rate of gas generation will occur.

2.5.5.7 Inoculum addition

Many researchers suggested adding inocula as a bioreactor management alternative. Municipal sewage sludge, animal manure, septic tank sludge and old MSW have been recommended as potential inocula. The effect of sludge addition on the MSW biodegradation is covered by Leuschner (1982), Pacey (1989) and Warith (2002). They concluded that the addition of sewage sludge has both a positive and a negative effect on the MSW biodegradation and methane generation. The positive effects of sludge addition occur after the methanogenic bacteria are already established and the landfill environment is optimum (pH neutral) for methanogenic bacteria (Christensen et al., 1992). This positive effect can be attributed to the following factors: 1) sludge can be a source of nutrients and active methanogenic bacteria, and; 2) sludge increases the moisture content. Similarly the negative impact of sludge addition to fresh waste is attributed to the acid accumulation that is associated with it which decreases the pH and inhibits the methanogenic bacteria (Barlaz et al., 1990).Anaerobically digested sewage sludge can serve as a seed to microorganisms as well as source of nitrogen, phosphorous, and other nutrients (Warith 2005). Gulec et al., 2000 reported that in 10 L laboratory-scale batch digesters filled with 2- year old MSW at ratios of 1:9, 1:6 and 1:4 (anaerobically digested sludge to waste on wet basis), pH of leachate ranged from 7.0 to 8.5 compared to sharp drop in pH levels to the acidic range in the control reactors (no sludge addition). This may be explained by the buffer capacity of sludge. On the other hand, Barlaz et al. 1987 observed carboxylic acid accumulations and decreases in pH associated with sludge addition to fresh MSW. The results of this study confirmed that sludge addition without buffer addition did not stimulate methane production. Moreover, Christensen, and Kjeldsen (1992) suggested that sewage sludge addition to MSW might have a limiting effect on waste biodegradation if the anaerobic conditions are already established. Furthermore, Erses and Onay (2003) suggested that the utilization of external leachate recycled from old landfills having desired acclimated anaerobic microorganisms, low organic content and higher buffer capacity into a young landfill could be a promising leachate management strategy for faster waste stabilization.

2.5.5.8 Particle size of waste

The use of MSW with a reduced particle size relative to unprocessed MSW provides a more homogenous waste. Ham and Bookter (1982) found that the shredding of waste increases the rate of decomposition and methane production. The well mixed shredded waste permits greater contact between the key refuse constituents required for methane production: moisture, substrate, and microorganisms (Barlaz et al. 1990). Waste shredding could lead to rapid oxygen utilization, increase rate of waste decomposition, and lead to early methane production (Ham and Bookter 1982; Otieno, 1989). Warith et al. 2005 indicated that shredded MSW produces leachate with higher peak COD concentrations and slightly lower minimum pH levels than unprocessed MSW. However, too small particle sizes could cause rapid waste hydrolysis, and lead to a build-up of acidic end products, that will have a negative impact on methane production.

Buivid et al. (1981) reported that waste shredding to particle size in the range of 250 to 350 mm particle sizes produced 32% more methane after 90 days than MSW with 100 to 150 mm particle sizes, and 100-150 mm shredded MSW produced 16 times more methane than a finely shredded MSW of less than 25 mm particle size. This is due to the fact that the smaller particle size increases the rate of hydrolysis and acid formation which in turn decreases the pH and postpones the production of methane.

Sponza et al. (2005) reported that the shredding of MSW has a positive impact on the degradation in anaerobic bioreactors with leachate recycle. They compared three types of reactors. The first reactor was loaded with raw waste, the second with shredded waste, and the third with compacted waste. At the end of the experiments (57 days later), they found that the reactor with waste shredding had the lowest COD and VFA concentrations and the highest methane percentage.

2.5.5.9 Lift design, daily cover and compaction of waste

The enhancement of waste biodegradation in the landfill is also affected by the lift thickness, daily cover and compaction of waste. The lift thickness has an adverse effect on the biodegradation of waste. Ham et al. (1982) found that the cell with a 2 m deep lift produced higher leachate concentrations and took a longer time to stabilize than the cell with a 1.2m deep lift. By doubling the lift depth from 1.2 to 2.4 m, the concentrations of leachate and stabilization time are doubled as well.

Stegmann (1983) suggested that the first layer should be uncompacted, so readily degradable organics can decompose aerobically and are allowed to stabilize before addition of subsequent lifts. Reinhart et al. (2002) indicated that the increased MSW compaction not only reduces waste ability to move moisture through waste but also makes the waste achieve level of saturation with less moisture addition because waste hydraulic conductivity is inversely related to waste density.

In order to minimize ponding and horizontal movement, Reinhart and Townsend (1998) suggested using of high permeability soils and/or alternative daily cover. 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 landfill space and cost saving, increase of waste hydraulic conductivity within the landfill and extended life of the leachate drainage layers efficiency (Wiles and Hare 1997).

2.5.5.10 Inhibitors

There are a number of compounds that can inhibit the biodegradation of solid waste besides O_2, H_2, acidic pH and high concentrations of heavy metals. These inhibitors are carbon dioxide, sulphate, and high concentrations of cations such as sodium, magnesium, and ammonium. The CO_2 acts as an inhibitor by raising the redox potential which has an effect on the acetic acid conversion to methane (Christensen et al., 1996).

The anaerobic ecosystem is considered to be rather sensitive to inhibitors. Researchers have reported many inhibitors of anaerobic degradation, e.g. oxygen, carbon dioxide, hydrogen, proton activity, salt ions, sulphide, heavy metals, and specific organic compounds (Christensen and Kjeldsen, 1989) .Cations such as sodium, potassium, calcium, magnesium and ammonium have been observed to stimulate anaerobic decomposition at low concentration while inhibit it at high concentrations (Christensen and Kjeldsen, 1989). High sulphate concentration can inhibit methane generation. It has been speculated that CO_2 acts as an inhibitor through the raising of the redox potential (Hansson,1982) , or 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).

2.5.5.11 Oxygen and hydrogen concentration

The activities of methanogenic bacteria are sensitive to the presence of oxygen. Extensive gas recovery pumping may create a substantial vacuum in the landfill, forcing to fill with air in 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). In reality, the oxygen that diffuses from the atmosphere into the landfill is consumed by aerobic bacteria in the top layers of the landfill .Aerobic bacteria in the top of the landfill, under normal condition, will cause solid waste to readily consume the oxygen and limit the aerobic zone of the compacted waste (Warith, 2003). The fermentative and acidogenic bacteria produce hydrogen whereas the methanogenic bacteria use the hydrogen as a substrate to produce methane. Propionic acid and butyric acids are the byproducts of acidogenic processes. The conversion of propionic acid requires a hydrogen pressure lower than $9X10^{-5}$ atmospheres (Christensen et al., 1989). A low partial pressure of hydrogen is required for the acidogenic processes. An increase in the partial pressure of hydrogen causes the generation of propionic and butyric acids with no further conversion, resulting in an accumulation of volatile organic acids which reduce the pH and inhibit the methanogenic bacteria (Christensen et al., 1989).

2.5.5.12 Pre-treatment

The degradation process can be enhanced with the pre-treatment of MSW. The process will enhance the acidogenic stage and decrease the accumulation of organic acids. This method is based on the stabilization of part of the waste through aerobic processes which will dilute the organic acids and cause a balance between the acidic phase and the methanogenic bacteria (Ham et al., 1982; Stegmann, 1983; Beker, 1987). The authors found that by placing fresh waste on top of the composted waste layer caused a shorter acidogenic stage and enhanced the methanogenic stage.

Chapter 3

Materials and methods

3.1 Introduction

This chapter describes the sample collection, preparation, laboratory tests and procedures and experimental program used in the current work. A detail experimental program was designed to study the effect of compaction on the hydraulic parameter of MSW. First, the detail investigation of compaction with hydraulic parameters was performed. After estimating certain optimum density range from hydraulic parameters, the effect of compaction on the degradation was performed at those particular densities. An experimental program was developed based on few densities and the degradation was monitored to study the effect of compaction on the degradation. Sample preparation procedures, design of devices and monitoring methods were explained subsequently.

3.2 Collection and storage of waste samples

Fresh solid wastes were collected from the working phase of the City of Denton landfill at three random locations, in three month intervals. The MSW samples were extracted using a backhoe loader, mixed thoroughly and quartered on the surface as shown in Figure 3-1(a, b and c). These samples were used to evaluate different properties of MSW. In the beginning 6 bags were collected and designated as waste-A. Similarly 10 bag of waste samples were collected on the second time and designated as waste-B. After collecting the MSW samples, the samples were taken to the laboratory and preserved at about 4°C in an environmental growth chamber. The sample collecting procedure from field, and storage in cold chamber are shown in Figure 3-1.

(a)

(b)

(c)

(d)

Figure 3-1 Waste collection and storage (a) Mixing of the sample (b) Samples divided into four quarters (c) Collection of wastes (d) Storage of the waste samples

3.3 Determination of general characteristics of fresh waste

The collected samples were stored in cold chamber at 4°C to preserve the moisture content. After opening the waste bags, general physical characteristics of the solid waste were found which include physical composition, moisture content, and maximum dry density. The samples were utilized to perform compaction effect on hydraulic parameter and decomposition.

3.3.1 *Moisture content*

The moisture content of the 16 MSW samples on both a dry and wet weight basis was determined according to the ASTM D-2216. After bringing waste samples in laboratory, moisture content of each bags were performed. Approximately 2 lb of samples were dried at 105°C in the oven for 24 hours and measured for moisture loss. Figure 3.2 shows sample inside the oven for the determination of moisture content. Moisture content (w_w) on a wet weight basis was determined using equation 3.1.

$$w_w = \frac{M_w}{M_t} \times 100; \ \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \ (3.1)$$

Similarly moisture content (w_d) on dry basis was determined using:

$$w_d = \frac{M_w}{M_s} \times 100; \ \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \ (3.2)$$

Where m_w is the mass of water, m_t is the total wet mass of MSW and m_s is the dry mass of solid MSW.

Similarly, moisture contents of all tested samples were measured based on equation 3-2 after performing permeability tests on all waste samples.

Figure 3-2 Samples placed in oven for determination of moisture content

3.3.2 Composition

The physical composition 16 bags fresh MSW samples was determined by manually sorting the waste components into the following nine categories: paper, plastic, food waste, textile, wood & yard waste, metals, glass, styrofoam & sponge, and others. The “others” category basically included soil, rocks, and fines that were difficult to manually segregate with observation. The Figure 3-3 shows one of the samples after sorting. The sorted components were then weighed individually, and weight percentages were determined.

Figure 3-3 Sorting of different components of MSW

3.3.3 Maximum dry unit weight/density

The maximum unit weight of Waste-A and Waste-B samples were determined according to standard proctor compaction method (ASTM D698). Six samples for waste-A and for waste-B were prepared. Waste samples were prepared according to composition of waste-A and waste-B and shredded into maximum 2.5 inch size. The shredded samples were prepared and dried. Moisture content was maintained from 10% to 110% in 20% interval. After preparing the samples leachate were added on the prescribed amount and mixed uniformly. A larger compaction mold was used to perform the standard proctor test. The mold has a 6 inch inside diameter and volume of 1/10 cubic feet. A 5.5 lb hammer was dropped 75 times for a fall height of 12 inch on each of three layers to achieve required compaction. The 75 blows instead of 25 was calculated based on the compaction energy per volume.

Energy transferred in standard proctor test, $E = n \times h \times (\frac{P}{V})$ …………… (3.3)

Where, n = number of blows, h = fall height, P = weight of hammer and V = volume of the mold. P and h are equal for the both molds.

For the normal sized mold, $E_1 = p \times h(\frac{n_1}{V_1})$

For the large sized mold, $E_2 = p \times h(\frac{n_2}{2})$

For E to be same in both cases, E1=E2

$$n_2 = n_1(\frac{V_2}{V_1})$$

$$n_2 = 25 \times (\frac{1/_{10}}{1/_{30}})$$

$$n_2 = 75$$

The dry unit weight of waste was calculated using the equation 3-4.

$$Unit\ Weight = \frac{Weight\ of\ Compacted\ Waste\ inside\ mold}{Volume\ of\ Mold}$$ ……..……….. (3.4)

Figure 3-4 Sample compacted for maximum dry unit weight determination

3.4 Experimental research activities

The ultimate goal of the current study was to observe the effect of compaction on the hydraulic parameter such as permeability and porosity as well as on degradation. The composition and compaction on the waste are also connected to degradation so that it has been focused on the interconnection of hydraulic conductivity to the degradation and estimating optimum density based on hydraulic parameters and degradation. Hydraulic conductivity is one of the most important parameter which is basically affected by density, composition and degree of saturation. In the current study the effect of compaction on the hydraulic properties of MSW had been focused at various densities and 3 densities were selected to observe the effect of compaction on the degradation of municipal solid waste in the laboratory. The flow chart for the current research activities are described in the Figure 3-5.

In the beginning, constant head permeability tests were performed on saturated waste samples at different compacted densities. Different types of porosities such as total, retained and drainable porosity were measured. Saturated water content & water retention capacity were also calculated for all various samples. Based on the effect of compaction on hydraulic parameters, few densities were selected to study the effect of compaction on the degradation and gas generation of solid waste were monitored for one year. Three density ranges were selected to observe the effect of compaction level. Optimum range of dry density were proposed based on the analysis of hydraulic conductivity, retained porosity, drainable porosity, leachate generation and gas generation criteria.

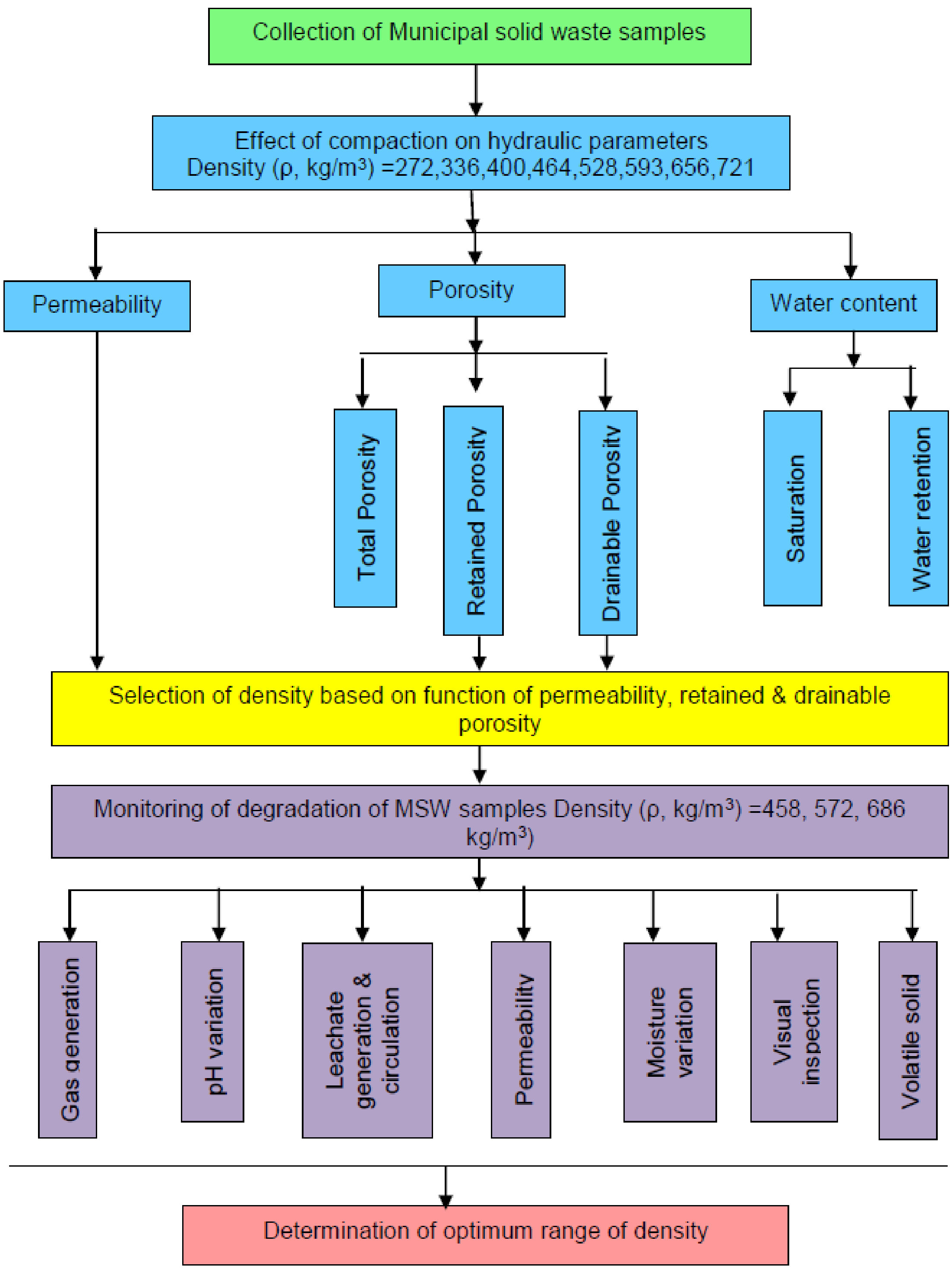


Figure 3-5 Flowchart diagram in the current experimental study

3.4.1 *Compaction effect on hydraulic parameters*

The effect of compaction on the MSW had been studied in three different ways.

- Effect of compaction on hydraulic properties in short term
- Effect of compaction on long term hydraulic conductivity

It was not possible to do tests on various composition of waste due to the limitations of equipment. Two different compositional variations on wastes were considered for the experimental study. The waste–A and waste–samples were prepared according to the compositions described in Table 3-1.

Table 3-1 Composition of the waste samples

Waste components	Waste-A (% by wt)	Waste- B (% by wt)
Paper	33	41
Plastic	28	18
Food	0.1	3.6
Textile	4.6	2.1
Wood/yard	10	6.4
Metal	2.0	3.4
Glass	0.3	0.5
Styrofoam/sponge	2.4	3.1
C&D	7.6	0.7
Fine contents and soil	12	21

3.4.1.1 Effect of compaction on hydraulic properties (short duration tests)

Different properties such as hydraulic conductivity, porosity and moisture content were determined from tests. Shredded waste-A; shredded waste-B and unshredded waste-B were tested to find out the hydraulic properties at different compaction level. Three different sizes of permeameter were utilized for both shredded wastes. In general, for municipal solid waste, there is no rational basis in literature for selecting a specific apparatus size (Hossain and Gabr 2009). Athanasopoulos (2008) recommended 1/6 the size of specimen (diameter or side) for bulky constituents (e.g. wood, gravel, glass) whereas 1/4th the size (diameter or side) of specimen for softer, easily folded, high aspect ratio constituents (e.g. paper, plastic). Additional research work is needed to identify the effects of size of the components compared to specimen size (Athanasopoulos 2008) Powrie and Beaven (1999); Beaven (2000) conducted a constant head flow test in a Pitsea compression cell (2 m in diameter and up to a 3 m height) to determine the permeability of crude unprocessed household waste. Pitsea cell was used to get a representative result from highly heterogeneous samples. Reddy et al 2009(b) used 6.3 cm and 30 cm diameter permeameter to perform permeability tests.

Different components of the fresh MSW were separated and stored. Individual components were mixed in prescribed amount according to Table 3-1 to prepare the similar type of waste samples. Un-shredded waste samples were considered only in large permeameters while shredded samples were considered for all size permeameters. It was not possible to compact the entire waste particles in all size permeameters therefore the waste was shredded to accommodate into the permeameter size. Larger particles of the collected waste were broken down into small size and some unbreakable big particles were discarded. The ratio (R) of the diameter of the permeameter to the particle size for the shredded waste was always maintained at approximately 2.5 in all permeameters. The purpose of shredding to provide uniformity in the mixing process. All the shredded components were mixed uniformly. The shredding of the waste, preparation of the sample, mixing of waste and devices are shown in the Figure 3-6, Figure 3-7, Figure 3-8 and Figure 3-9.

Figure 3-6 Mixing process of shredded waste samples for small and medium permeameter

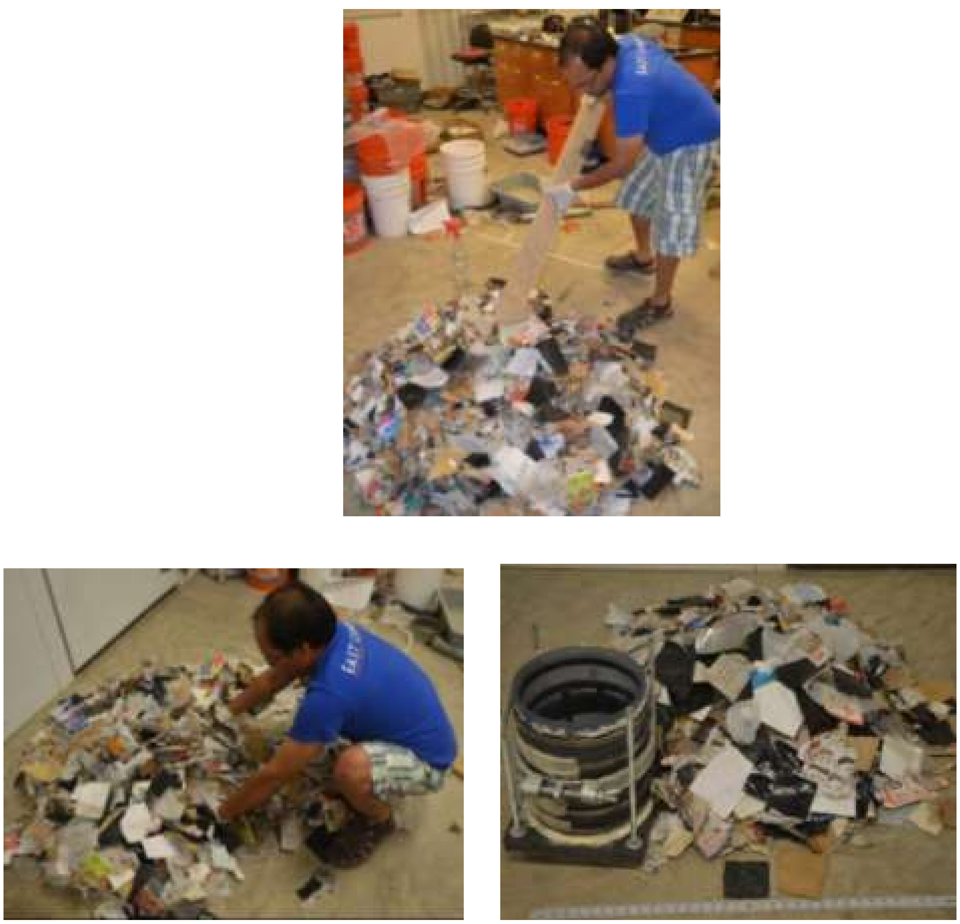


Figure 3-7 Mixing procedure of the shredded waste for large size permeameter

Figure 3-8 Different size shredded samples, 10.2, 6.4 & 2.5 cm

Similarly un-shredded samples were tested only in large 25.4cm diameter permeameters. Similar technique was followed to prepare the sample (Figure 3-9). The main purpose of preparing the sample with the exact weight of components was to make consistency in the testing. In these tests, wet waste components were taken initially but the whole sample was dried slowly at hot chamber at 100° F to get the exact dry weight of waste.

Figure 3-9 Preparation of shredded MSW sample

3.4.1.2 Effect of compaction on long term hydraulic conductivity

The sample preparation was similar in each case but only one variety of waste samples was prepared to perform long duration permeability test. The tests were performed on waste-A at three different dry density. To prepare the similar waste composition of the waste samples, individual components were taken in respective amount and mixed together. Constant head permeability test was performed to measure the hydraulic conductivity every day for 116 days. Large 25.4 cm diameter permeameter was used to perform these tests. The ratio (*R*) of diameter of permeameter to particle size was maintained approximately 2.5. The waste sample was prepared by shredding waste components which were greater than 10.2 cm size. The sample was compacted using the tensile compression machine in order to get targeted density. Dry densities of samples were 444 kg/m^3, 571 kg/m^3 and 696 Kg/m^3. The initial moisture content of waste samples was approximately 26.5% while doing the compaction.

3.4.2 Experimental design for compaction effect on hydraulic parameters

An extensive laboratory investigation was conducted to determine hydraulic characteristics of the waste at various dry densities. Dry densities were approximately varied from 272 kg/m3 to 721 kg/m^3 for different wastes. The details of the experimental program are presented in Table 3-2.

Table 3-2 Experimental program for compaction effect on hydraulic parameters

Approx. Dry density (kg/m^3)	Waste type	Permeameter (size, cm)	No of sample	No of tests		
				Permeability	Porosity	Moisture Holding Capacity
272	Fresh Shredded	6.35, 15.24, 25.4	2	6	6	6
	Fresh Un-shredded	25.4	1	1	1	1
336	Fresh Shredded	6.35, 15.24, 25.4	2	6	6	6
	Fresh Un-shredded	25.4	1	1	1	1
400	Fresh Shredded	6.35, 15.24, 25.4	2	6	6	6
	Fresh Un-shredded	25.4	1	1	1	1
464	Fresh Shredded	6.35, 15.24, 25.4	2	6	6	6
	Fresh Un-shredded	25.4	1	1	1	1
528	Fresh Shredded	6.35, 15.24, 25.4	2	6	6	6
	Fresh Un-shredded	25.4	1	1	1	1
593	Fresh Shredded	6.35, 15.24, 25.4	2	6	6	6
	Fresh Un-shredded	25.4	1	1	1	1
656	Fresh Shredded	6.35, 15.24, 25.4	2	6	6	6
	Fresh Un-shredded	25.4	1	1	1	1
721	Fresh Shredded	6.35, 15.24, 25.4	2	6	6	6
	Fresh Un-shredded	25.4	1	1	1	1

Similarly, constant head permeability tests were conducted to investigate the variation of permeability with time. The tests were performed at three different unit weight of waste-A samples. The experimental program are presented in Table 3-3

Table 3-3 Experimental program to study variation of permeability with time

Variable		Series of Experimental set up	Comments
Dry density (kg/m^3)	Dry density (pcf)		
444	27.5	1	Monitoring permeability every day
571	35.4	1	
696	43.5	1	

3.4.3 Effect of compaction on the degradation

Three density were selected for this study based on effect of compaction on the hydraulic conductivity. The tests were performed on waste-B at three dry densities as 458 kg/m^3, 572 kg/m^3 and 686 Kg/m^3. Waste with similar composition was utilized to monitor the degradation tests. Sample preparation and compaction process were similar to the sample preparation for permeability tests. The initial moisture content of waste samples was approximately 40% while doing the compaction. The ratio of diameter of permeameter to particle size was maintained approximately 2.5. The sample was compacted in various layers in order to get targeted density. The initial moisture content of waste samples was approximately 40% on the dry basis while doing the compaction. Effect of compaction on the biodegradation of MSW was monitored for 1 year. The apparatuses were equipped with all necessary units required to those for bioreactor cells such as leachate recirculation; gas collection, and leachate collection systems. Along with these set up hydraulic conductivity of the compacted MSW were also carried out in each month to see the degradation level. The details of the experimental program are presented in Table 3-4.

Table 3-4 Experimental program to study effect of compaction on the degradation

Variable		Series of Experimental set up	Comments
Dry density (kg/m^3)	Dry density (pcf)		
458	28	1	Monitoring gas generation, pH variation, leachate generation, variation of permeability every month, leachate recirculation, volatile solids
572	35	1	
686	42	1	

3.5 Test method and procedures

Hydraulic conductivity, porosity and moisture retaining capacity of the solid waste were measured for different densities and composition of wastes. Maximum dry density were also determined through standard proctor test. Depending on the results of permeability versus dry density, three densities were selected to observe the compaction effect on degradation and gas generation.

3.5.1 Determination of hydraulic conductivity

The saturated hydraulic conductivity of MSW samples was determined from constant head test using three different size permeameters. However, due to lack of standard test procedure for determination of permeability of MSW, the standard test procedure for determination of permeability of the granular soils (ASTM D 2434-68) was adopted for the current research work. The hydraulic conductivity tests were performed all waste–A, waste–B and un-shredded waste samples. Permeability conductivity tests were performed at various dry densities for shredded waste–A; shredded waste–B and un-shredded waste respectively. The variations in hydraulic conductivities at all densities for all samples were recorded and analyzed to evaluate optimum compaction range. The same procedures were followed while using all available permeameters. The coefficients of permeability of all wastes were estimated using the equation 3-5.

$$k_s = \frac{v \times L}{a \times h \times t} \ldots\ldots\ldots\ldots\ldots\ldots (3.5)$$

Where k_s= coefficient of saturated permeability; v= Volume of liquid; L= Length of Sample; t= Time of water collection; h= Head of liquid; a= Area of sample

3.5.1.1 Monitoring of hydraulic conductivity with time

Long term permeability was monitored in two waste samples. Three different unit weights were selected which are described in Table 3-4 and Table 3-5. In waste-A samples hydraulic conductivities were monitored for 116 days. Similarly, constant head permeability tests were performed on waste-B along with measuring generated gas from bioreactor cells. The coefficients of permeability were measured on the monthly basis to see the effect of degradation. Gas was appeared while performing long duration permeability tests. Hydraulic conductivity tests were performed by removing all the entrapped air that was generated due to degradation process.

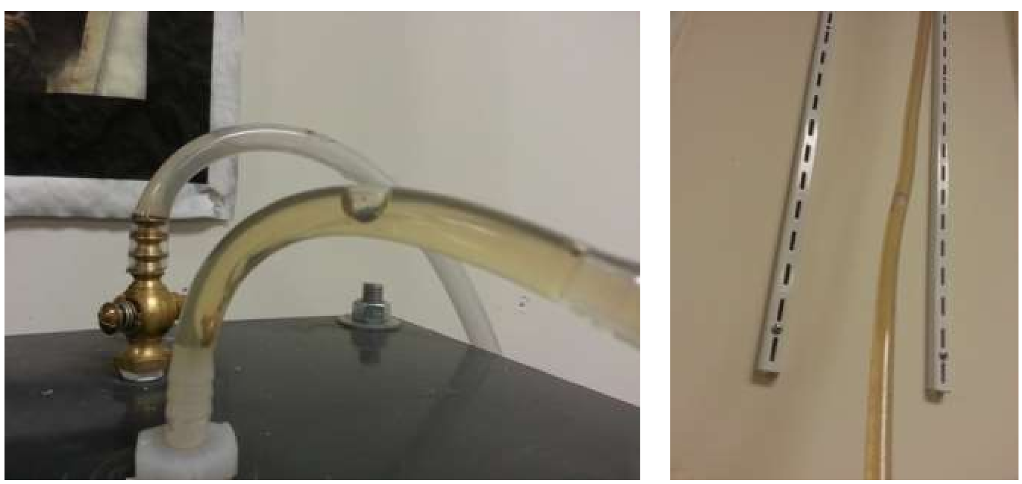
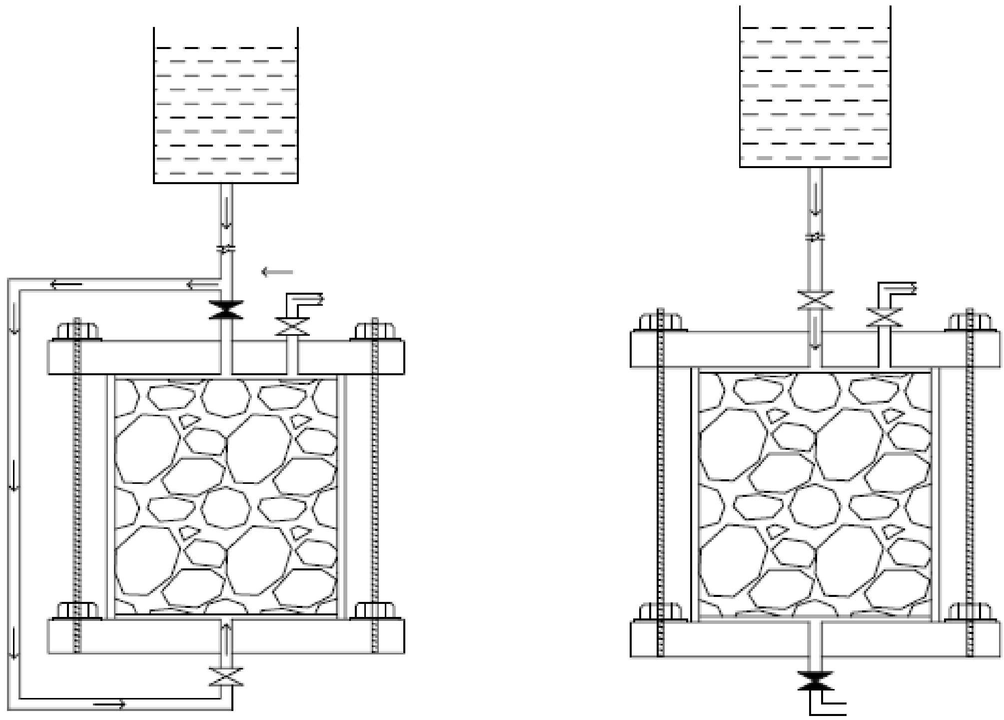


Figure 3-10 Accumulation of gas bubbles on pipe and moving upward

Due to continuous supply of water degradation took place on the permeameter and bioreactor cells. The ongoing degradation process resulted gas which was the main factor reducing the permeability over time. As the gas produced, the space was occupied with the gas and the hydraulic conductivity decreased significantly. Gas bubbles were observed in the pipe and moved upward while performing test which are shown in Figure 3-10. In order to remove the gas from pipe and permeameter, the gas valve was released in two ways as explained in Figure 3-11. In the first process, the top inlet water pipe was kept closed while gas vent and bottom water pipe was kept opened and water was applying from bottom of the cell into the waste sample to remove the entire accumulated air inside the sample and geocomposite layer. This process was able to remove the entire air stored in the waste sample (Figure 3-11 (a)).

In the second process, the top inlet water pipe and gas valve were kept opened while the bottom outlet was kept closed to just remove the air accumulated on top geocomposite layer and on pipe. In this process water was flowed continuously from water tank through top water inlet pipe and discharged through gas vent. Since the water flowed from top part of the sample which cannot remove the entire air stored in the waste sample but only accumulated gas on the top of the waste sample (Figure 3-11 (b)). While performing long phase permeability tests, generated gas was released in both two ways for waste-A whereas entire gas was released from first process for waste-B.



(a) (b)

Figure 3-11 Gas release process (a) Water applying from bottom of the sample (b) Water applying from the top of sample

3.5.1.2 Hydraulic conductivity without time variation

Constant head permeability tests were performed on shredded and unshredded 51 waste samples. Waste samples were first saturated by flushing tap water from the bottom of samples in order to remove entrapped air from waste. Before performing the tests, gas was released from the gas vent if any gas was generated. After complete saturation, constant head permeability tests were performed at various dry densities for shredded waste–A; shredded waste–B and un-shredded waste, respectively. The detail experimental program is already explained in Table 3-2. The hydraulic properties of all waste samples were determined using 3 different size permeameter as shown in Figure 3-20. A ratio (*R*) of 2.5 were maintained for all shredded waste samples between diameter of the permeameter to the size of the waste particle. The shredded waste particles were shown in Figure 3-8.Similarly constant head permeability tests were carried out utilizing un-shredded waste. Larger diameter permeameter was only used for unshredded waste.

3.5.2 Determination of porosity

Waste samples were saturated by applying tap water from the bottom of samples under the same hydraulic head of 190.5 cm for 24 hours. Before taking the weight of saturated weight of samples with permeameters, gas was released from the gas vent if any gas was generated in order to fill all the void space with water. After saturation, the weights of all saturated samples were recorded for all dry densities 272 kg/m^3 to 737 kg/m^3 for all waste samples. The weight of the saturated waste was calculated according to equation (3.6).

Saturated wt. of sample (w_s) = Total wt. of saturated waste with device – wt. of device………………………………. (3.6)

After taking the weight of waste in saturated condition, the saturated wastes were allowed to drain under the gravity flow at room temperature for 24 hours. The weights of all samples after 24 hours were recorded for all samples. The weight of the waste at stable condition was calculated according to equation (3.7).

Wt. of sample in stable condition (w_{rw}) = Total wt. of waste sample in stable condition with device – wt. of device…………… (3.7)

The same procedures were followed in all permeameters. Different equations were utilized to calculate different type of porosities. Basically, the volume of pore space divided by the volume of waste sample determines its porosity. This measurement is given as a percentage of the volume of the void in waste. In waste materials, porosity can be categorized in three ways as total porosity, effective porosity and drainable porosity.

3.5.2.1 Total porosity

If the waste material is fully saturated all the void are assumed to be occupied with water. The weight of water can be found by subtracting saturated weight of waste to dry weight of waste. The volume occupied by the water is the ratio of the weight to the unit weight of the water. The ratio of volume of water to the total volume of waste is called total porosity of the waste. Basically it is the volumetric water content of the waste in the saturation state. In order to estimate the total porosity, initially the dry sample in the cell was saturated and weight was recorded. The saturated weight of the samples can be calculated using the equation (3.6). Weight of water was calculated using equation (3.8). The increase in weight over the dry weight will represent total water absorbed by the samples. It was assumed that all void was occupied with water upon saturation process. The volume of void was calculated using the equation (3.9). After knowing the volume of void and total volume of the sample, the total porosity can be calculated using the equation (3.10).

$$w_w = w_t - w_s \qquad (3.8)$$

$$v_v = \frac{w_w}{\gamma_w} \qquad (3\text{-}9)$$

$$n_t = \frac{v_v}{v_t} \qquad (3\text{-}10)$$

Where, w_w is the weight of absorbed water by waste, w_t is the weight of saturated waste, w_s is the weight of dry waste, v_v is volume of total void and v_t is total volume of waste sample, γ_w is the unit weight of water(9.81kN/m^3).

3.5.2.2 Retained/Effective porosity

If a fully saturated waste material is allowed to drain under gravity, the waste will lose water and ultimately reach a stable condition which is called as field capacity. The ratio of volume of remaining water at field capacity to the total volume of waste is called retained/effective porosity. Basically it is the porosity occupied with water inside the waste. In order to estimate the retained/effective porosity, the saturated waste sample was allowed to drain for 24 hours under the gravity flow and the sample attained a stable condition. The moisture retaining capacity at this stage is also called the total absorptive capacity of waste. The volume of pore space (v_w) occupied with water in the total absorptive capacity was calculated using the equation (3.11) and effective/retained porosity (n_e) can be calculated using the equation (3.12).

$$v_w = \frac{w_{rw}}{\gamma_w} \dots\dots\dots\dots\dots\dots\dots\dots\dots\dots \text{(3.11)}$$

Where, v_w is the volume of water at stable condition after 24 hours of gravity flow, w_{rw} is the weight of the remaining water inside the waste after 24 hours of gravity flow.

$$n_e = \frac{v_w}{v_t} \dots\dots\dots\dots\dots\dots\dots\dots\dots\dots \text{(3.12)}$$

3.5.2.3 Drainable porosity

When a fully saturated waste material is allowed to drain under gravity flow, the water content of waste will decrease. The void inside waste will be occupied with water and with air. The waste cannot remain always in saturated condition. Waste has certain capacity to hold moisture and after exceeding this limit it start to drain excess water from its pore. The volume of freely draining water per unit volume of waste defines as the drainable porosity. Theoretically the difference between total porosity and effective/retained porosity gives drainable porosity. The weight of draining water (w_d) was calculated using the equation (3.13). The volume of drainable water was calculated using the equation (3.14).

$$w_d = w_w - w_{rs} \dots\dots\dots\dots\dots\dots\dots\dots \text{(3.13)}$$

$$v_{dw} = \frac{w_d}{\gamma_w} \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \text{(3.14)}$$

Where, v_{dw} is the volume of draining water. After calculating the volume of drainable water from the waste, the drainable porosity (n_d) was calculated using the equation (3.15).

$$n_d = \frac{v_{dw}}{v_t} \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \text{(3.15)}$$

Where, v_{dw} =volume of drainage water after 24 hours gravity flow; and $v_t =$ total volume of waste

3.5.3 Determination of variation of water contents

Municipal solid waste has capacity to hold moisture which is an important factor to enhance the degradation. There are many factors which can affect moisture holding capacity for any solid waste. Moisture balance is a vital process to maintain the moisture content inside the landfill. In order to understand the moisture balance phenomena inside the landfill, it is necessary to have an idea how the moisture holding capacity is affected by the density. This research had also been focused on the effect of density on the moisture retaining capacity. The moisture content can be defined in the literature in three different ways: dry gravimetric moisture content, wet gravimetric moisture content, and volumetric moisture content (Sharma and Reddy, 2004); however, in this study moisture content is defined as dry gravimetric moisture content and is calculated using the equation 3.2.

3.5.3.1 Moisture retention capacity

After taking the saturated weight of waste sample, waste material was allowed to drain under gravity for 24 hours. Its water content will decrease as drainable pores empty and eventually reach a stable state (termed as the field capacity) when no further drainage occurs. The weight of all drained samples was recorded for all wastes. The moisture content was calculated using the equation (3.2) for all drained samples.

3.5.3.2 Saturated moisture content

After saturation, the weights of all saturated samples were recorded for all dry density samples for waste–A, waste–B and unshredded waste-B, respectively. The total absorbed water was calculated using the equation 3.6. The saturated moisture content was calculated using the equation 3.2 after knowing the amount of total water absorbed by waste samples.

3.5.4 Monitoring of bioreactor cell operation

One of the major task of this research was to observe the effect of compaction on the degradation of the municipal solid waste. The experimental program was described in Table 3-4. Laboratory scale simulated landfill bioreactor cells were designed based on the results obtained from permeability tests. This task included operating and monitoring bioreactor cells for 1 year duration. This step involved measuring parameters gas volume and percentage of methane (CH_4), carbon dioxide (CO_2) and oxygen (O_2) in gas; and measuring leachate volume and pH, recirculation capacity of leachate, variation of permeability with degradation, volatile solid contents, and visual inspection of the degradation label based on color.

3.5.4.1 Bioreactor gas collection and measurement

The volume of the gas production was measured by emptying the gas collection bags with an air sampling pump (Universal XR pump Model 44XR by SKC) connected to a calibrator to get a gas pumping rate. Time was recorded through the stop watch. The total volume of the gas generation was estimated by multiplying the rate of pumping and time to empty gas bags. LANDTECGEM 2000 was used for measuring percentage of Methane (CH_4), Carbon Dioxide (CO_2), Oxygen (O_2), and other gases, respectively. The frequency of gas sampling depended on the amount of gas generation in the bioreactor cells. During the initial stages of degradation, the gas generation was more so that bags were emptied more frequently to avoid excessive buildup in the gas bags. As degradation progressed, the rate of gas generation decreased and the frequency of sampling was reduced accordingly. The Figure 3-12 shows instruments used for gas volume and composition measurement.

(a) (b)

Figure 3-12 Gas collection (a) Measuring the composition of gas (b) Volume of gas measuring device

3.5.4.2 Leachate generation and recirculation

Leachate volume generated was recorded from bioreactors cells. Basically, the quantity of generation was mostly based on the recirculation quantity of leachate. The leachate will only generate if the moisture contents exceed the absorptive capacity of waste. In order to increase the moisture content above absorptive capacity, leachate was periodically recirculated. The recirculation greatly depended on the label of density. This is one of the indication of compaction label of the MSW. Generally the lower density waste, the waste can absorb huge amount of water in short period of time as compared to high density. During initial period of operation, the bioreactors received more amount of water which was dropped down in the compacted samples. If the leachate generation was very low, it was tried to recirculate in the same ratio to all reactors. Water was added if the leachate was insufficient for recirculation. Generally the leachate/water recirculation amount was very low and decreased after initial saturation in the more compacted samples. The leachate to be recycled was neutralized (pH≈7) with KOH buffer as necessary. Leachate collection and recirculation are presented in Figure 3-13 and Figure 3-14.

Figure 3-13 Leachate collection and measurement

Figure 3-14 Leachate recirculation

3.5.4.3 Variation on pH of leachate

After collecting the leachate from the drainage bag, pH was measured using a bench-top meter by OAKTON shown in Figure 3-15.

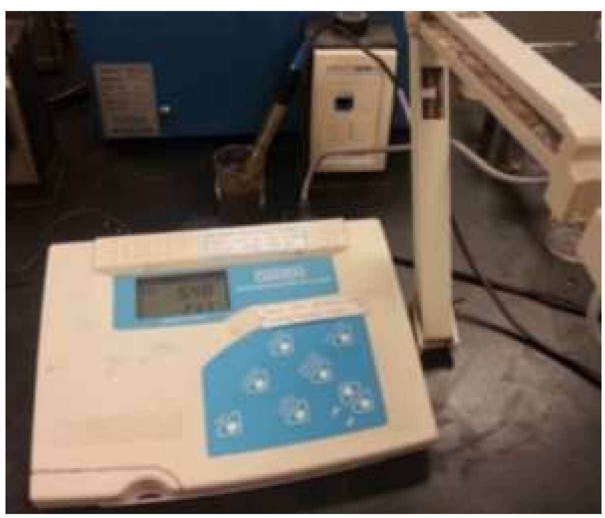


Figure 3-15 Measuring pH of collected leachate

3.5.4.4 Monitoring moisture content of degraded waste

The initial moisture content of the waste in each reactor was determined to understand the moisture content of waste. However, before installing each reactor, water were added to the waste and maintained at 40% on the dry weight basis and it referred to as initial moisture content of the reactor. Basically moisture content was found by taking approximately 2 lb of waste and the waste was dried in an oven at 105°C (±5 $_{o}$C) for 24 hours. Extra care was taken while finding the moisture content of samples containing higher percentages of food because it was reported that some of the organic matter from food waste volatilizes at 105°C (Angelidaki et al., 2009). Hence, food waste samples were dried at 65°C (± 5°C) for about 5-7 days, until the samples reached constant weight. Moisture content on a dry weight basis (w_w) was determined using the equation 3-1.

3.5.4.5 Visual inspection of degradation level

The color of the waste also provide the degradation label. Although exact label of degradation was not possible to find out, it gave a tentative idea of degradation label. Generally, fresh waste look very clear without any black color. All components looked very clean and clear in the beginning. The similar type of wastes were compacted in the bioreactor cell in the beginning. The change in color of MSW with degradation was also observed. As the bioreactor cells were transparent, the waste can be looked clearly outside. All the bioreactor cells were dismantled in the same time after 1 year. The color of all samples were compared.

3.5.4.6 Determination of volatile solids

The test solids procedure followed a modified version of Standard Methods APHA Method 2440-E. Volatile solids are an indicator of the organic content in the waste samples. Organic content of the waste is expected to decrease as the waste degrades. Initial volatile solid presented in waste samples were determined. Since the composition of each samples in reactors were same in the beginning, it was assumed the amount of volatile solid present in each reactor before degradation were same. After dismantling the reactors waste samples from each reactor were taken and performed composition tests. After finding the composition volatile solids were found for each samples. The volatile solids concentration in the degraded waste was also measured in two sample.

Dried waste samples were ignited in a muffle furnace at 550°C (± 10°C) for about 2 hours, or until it reached constant weight. The percent weight lost during ignition was the volatile solids in the waste. To find the content of volatile solid present in MSW, the samples were oven dried at 105°C temperature. These samples were then shredded into smaller pieces. About 50 grams of dried sample were taken in porcelain basin for each test and placed in the muffle furnace at 550°C for at least one hour to burn completely to ashes. Test setup and equipment used to measure volatile solid are presented in Figure 3-17. The loss in weight of sample due to burning was recorded and equation (3.16) was used to find the volatile solids.

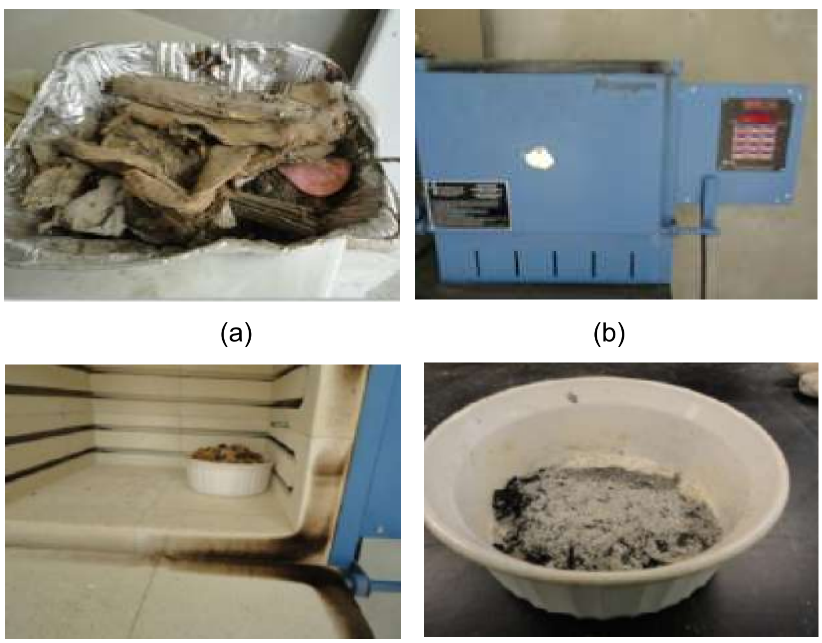
$$\text{VS (\%)} = \frac{w_l}{w_t} \times 100 \ldots\ldots\ldots\ldots\ldots \text{(3.16)}$$

Where, w_l is the weight loss in dry waste after burning; w_t= dry weight of sample before burning

Figure 3-16 Finely shredded waste samples prepared to grind in machine

Figure 3-17 Waste grinded in grinding machine for volatile solid test

Figure 3-18 Finely grinded waste samples ready for burning



(a) (b)

(c) (d)

Figure 3-19 Volatile solids determination (a) Oven dried sample, (b) Muffle furnace, (c) Shredded sample in the furnace, (d) Burned sample

3.6 Equipment used in research

Various size permeameters were used to perform the constant head permeability tests along with measuring porosity and moisture contents of the all prepared waste samples. Similarly laboratory scale bioreactor cells were designed to monitor the degradation of the waste samples.

3.6.1 Permeameters

The laboratory testing program consisted of determination of hydraulic conductivity, porosity, moisture content of fresh waste using variable size of permeameters. The hydraulic properties of the MSW samples were determined using three different permeameters, as shown in Figure 3-20. The small and medium-scale permeameters are available commercially and are used for coarse-grained soil. The large permeameter shown in Figure 3-20 (right) which was not available commercially so that it was designed specifically to test for solid waste. The reason for using the large permeameter was to keep bigger particle size while performing tests. Generally waste particles are generally large in the fresh stage and difficult to put inside permeameter without shredding into smaller particles. Three different sizes of permeameters were used to estimate the hydraulic properties of different wastes and to observe the variations with the sizes of permeameters. Permeability and porosity of fresh solid waste were estimated from different sizes of permeameters as shown in Figure 3-20.

Figure 3-20 Different size permeameters, small-size 6.35 cm; medium-sized 15.24 cm; large-size 25.4 cm diameter (from left to right)

3.6.1.1 Small 6.35cm diameter permeameter

A small-size rigid-wall permeameter was used to conduct constant head hydraulic conductivity tests in accordance with ASTM D 2434(ASTM 2006). This permeameter is generally used for granular soils in determining the coefficient of permeability via the constant or falling-head method for laminar flow of water. This permeameter was commercially available in the name as HM-3891. The sample diameter was 6.35 cm, height 15.25 cm, and weight of waste varied from 0.13 to 0.35 kg. Several testing samples were prepared at certain density intervals. All waste samples at various densities were tested for permeability and porosity. The maximum size of waste particle used in the small scale permeameter was 2.5 cm. The small scale permeameter is shown in Figure 3-20 (left).

3.6.1.2 Medium 15.24 cm diameter ppermeameter

A medium-size rigid-wall compactor permeameter was also used to conduct constant head hydraulic conductivity tests in the current study. This permeameter is generally used for sand and gravel type soil. The commercial name of the permeameter is H-4146. The sample diameter was 15.24 cm, height 15.5 cm, and weight varied from 0.76 to 2.0 kg. Several testing samples were prepared at certain density intervals to conduct tests using the medium-scale permeameter. All waste samples at various densities were also tested for permeability and porosity. The maximum size of waste particle used in the medium scale permeameter was 6 cm. The medium scale permeameter is shown in Figure 3-20 (middle).

3.6.1.3 Large 25.4 cm diameter permeameter

The Figure 3-20 (right) and Figure 3-21 shows the specially designed large-size rigid-wall permeameter. The diameter and height of the permeameter are 25.4, and 29.2 cm, respectively. This type of large permeameter was not available commercially so that it was designed specifically to conduct tests for waste materials. The permeameter was designed considering the particle size of MSW. PVC pipe and PVC plate were used to construct the large permeameters. Bottom plate was fixed at the pipe in order to make air tight and also making nonmovable while doing the compaction. While doing compaction, the pipe might move if there is high stress applied or if the pipe is tilted. Considering many of these factors, the pipe was fixed at bottom plate. The top plate was kept open in order to keep solid waste inside the cell. After filling the pipe with waste, top plate was kept on the pipe. The top plate had groove whose size was exactly equal to the thickness of the pipe. O ring was also used to make air and water tight during the tests. O ring was put in groove of the top plate and fixed with soft sealant. Top plate was kept above the pipe to fit exactly into the groove which insured perfect tightness. Generally, most of the permeability tests in waste were performed using the small-size permeameter designed for granular type soils. Waste material and its behavior are extremely different from soil, which might cause not representative values to the field while using the small-size permeameter. This kind of error can be minimized by using the large-scale permeameter. Several researches were designed a large equipment, which was called “Petsea Compression Cell”. Beaven and Powrie (1995), Powrie and Beaven (1999), Beaven (2000), Hudson et al. (2001), Hudson et al. (2004), Powrie et al. (2005), Powrie et al (2008) performed many hydraulic conductivity tests using the large Petsea Compression Cell. Generally, it is assumed that the data is more accurate and more representative to the field conditions when large equipment is used for waste. Basically, all components of variable sizes can be compacted in larger permeameters because of the large size of permeameter. The large size permeameter and its parts are shown in Figure 3-21.

Figure 3-21 Large-size assembled permeameter and its parts

3.6.2 *Design of bioreactor cells*

The design of the bioreactor was modified from the large size permeameter so that it had almost similar design except top cover. The top plate was replaced with different type of plate in reactors. The top plate had two opening (Figure 3-21) for permeameters while the top plate had four opening as shown in Figure 3-25(b). The schematic diagram of the bioreactor cell is shown in Figure 3-23. In Bioreactors, 4 symmetrical holes were made instead of two non-symmetrical holes. Two holes were used to circulate leachate and two holes were used to collect gas from the reactors. In bottom plate one hole was made at the center which was designed for leachate collection which was similar in bioreactor cell and permeameter. The volume of bioreactors were 0.5 ft^3. The bioreactor cells were modified for gas and leachate collection and for water addition (See Figure 3-24). Before filling the reactors with waste, all reactors were leak-checked (see Figure 3-22). Leak tests were conducted with filling reactors with water from water tank from the base of reactor. All holes at the top plate were kept closed in order to observe if there was any leakage. If there was leakage the water would flow from the reactors and the water table at the tank would lower down from the initial position. There was not any drop on the water table on the tank. To verify that the reactors were monitored for 1 day. The water label in the tank at 12 and 24 hours was recorded to confirm that it was no any drop in the water table. Reactors were then filled with waste components, as described in the Experimental Design section (Table 3-4). A 25.4 cm diameter piece of high drainage capacity fabric (Geocomposite) was placed at the bottom of the reactor and overlain with waste (see Figure 3-25 a). Geocomposite had the thickness of quarter inch which had geonet sandwiched between geotextiles. The purpose of using high drainage capacity geocomposite was to provide drainage layer on the bottom of reactors. The leachate generated from all over the waste flow towards the center for the collection system. Each reactor was placed in one of the constant room temperature in locations, and connected to a leachate collection bag (2-L Kendall-Ken Guard Drainage Bag) and gas collection bag (22-L Cali 5-Bond Bag, Calibrated Instruments, Inc.).

Figure 3-22 Leak-checked test on reactor

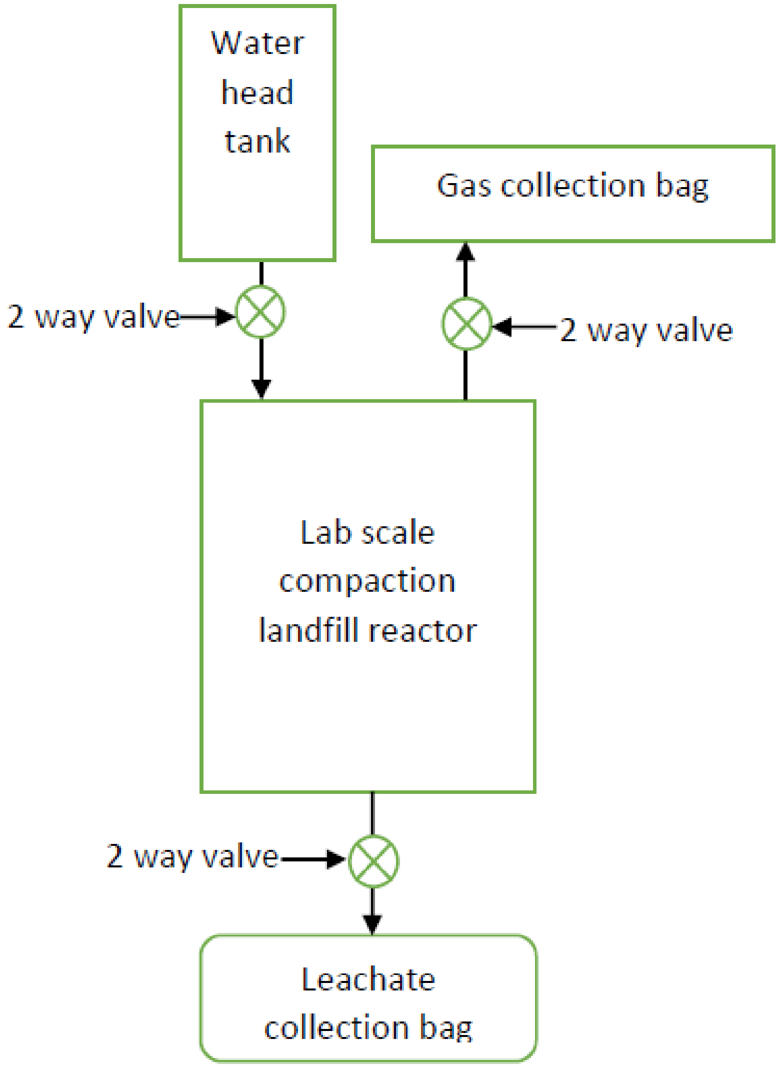


Figure 3-23 Schematic diagram of bioreactor cell with operation systems

Figure 3-24 A bioreactor cell with different operation systems as leachate collection, gas collection and leachate recirculation

(a) (b)

Figure 3-25 Showing the inside construction of bioreactor cell, parts and assemblage (a) Inside part of reactor (b) Top plate showing pipe connection

Figure 3-26 Three bioreactor cells under supervision

3.7 Compaction process of waste samples

First of all, the weight of samples were calculated based on the targeted density. Since, the volume of the molds were known, dry weight were calculated for each tests for all permeameters. Dry samples were taken to make consistency on the composition for all tests. The samples were compacted using the tensile compression machine as shown in figure 328. Samples were sprayed with 20-30% water and left for 3-4 hours before compaction in order to ensure uniform distribution of water to all components. The waste samples were compacted in several layers in order to make uniform density. Higher the density, the higher the number of compacting layers was considered. In high densities, the compaction of the waste created a high pressure on the wall of permeameter cell. In order to protect the permeameter from possible breaking and buckling, a strong big stiffener and small stiffener were used all over the pipe. The compaction steps are shown in Figure 3-28 Figure 3-29 and Figure 3-30 for small, medium and large size permeameter/bioreactor cell, respectively.

Figure 3-27 Tensile compression machine used for waste compaction

Figure 3-28 Compaction steps of MSW samples on small 6.35 cm permeameter

Figure 3-29 Compaction steps of MSW samples on medium 15.24 cm permeameter

Figure 3-30 Compaction steps of MSW samples in large permeameter and bioreactor cells

Chapter 4

Results and discussion

4.1 Introduction

In this chapter, the results obtained from the experimental study has been presented in detail. Fresh waste samples were collected from city of Denton landfill, Texas and brought to the laboratory to perform tests. The general characteristics of fresh waste were found before performing the compaction and degradation tests. The properties of any solid waste in the landfill are greatly influenced by factors such as compaction, composition. These properties are useful in designing and operating bioreactor landfill. The waste is the heterogeneous materials because of consisting of many types of waste components. The current study has been focused on the effect of compaction on the hydraulic conductivity, total porosity, retained porosity, drainable porosity, saturated moisture content and water retention capacity of waste.

Similarly degradation of the MSW is an important issue for bioreactor landfill. The degradation is influenced by many factors. The effect of compaction on the degradation of solid waste also had been studied as a part of research. The quality of leachate produced, gas generation, leachate generation and circulation, settlement or decomposition rate are highly dependent on the compaction level. The gas generated from the bioreactor was studied to determine the total methane production and composition. The effects of compaction on the leachate generation, circulation, pH variation, permeability variation were also studied.

The experimental results are presented and discussed in this chapter, which is divided into three sections. The first section includes the characteristics of municipal solid waste components (moisture content, physical composition and maximum dry density). The second section includes the effect of compaction on hydraulic parameters. The gas generation data from the laboratory scale landfill reactors, along with leachate volume, pH and probable moisture content inside the reactor, are presented in the third section.

4.2 Physical composition

Fresh solid wastes were collected from working phase from the City of Denton landfill. Solid wastes collected at 2 times on three of month interval were selected. There was slightly seasonal variation on the composition of the MSW so that those two different compositional variations were considered for this experimental study. The physical composition and amount of degradable materials of MSW samples was determined on a weight basis. In the first six bags were collected and found out the composition of each bags. The average of these six bags named as waste-A. Similarly, 10 bags were collected on the second time and composition tests were performed to find out the composition of each bags. The average of these 10 bags named as waste-B. The composition tests on each bags of waste- A and waste-B were described in Table 4-1 and Table 4-2, respectively. The average physical composition results on a weight basis are presented in Figure 4-1 and Figure 4-2 for waste-A and waste-B respectively. Based on the experimental results, it was found that paper was 32.9% on waste-A and 41.4% on waste-B on a weight basis which was the major component in both wastes. Besides paper, waste–A included 27.9% plastic, soil and fines content 12.0% yard/wood 10.2%, C&D 7.6%, textiles 4.6%, Styrofoam and sponge 2.4%, metal 2 %, glass 0.3%, food 0.1%. Similarly waste–B included fine particles and soil 21.1%, plastic 17.7%, yard and wood 6.4%, food 3.6%, metal 3.4%, Styrofoam and sponge 3.1%, textiles 2.1%, C&D 0.72%, glass 0.48%. The MSW samples of each bags were also categorized as degradable and non-degradable which are shown in Table 4-3 and Table 4-4 for waste=A and waste-B, respectively. The average amount of degradable and non-degradable contents for waste-A and waste –B are also shown in Figure 4-3 and Figure 4-4.

Table 4-1 Physical composition of waste-A by weight basis

Sample No.	Physical Composition (% by weight)									
	paper	plastic	food waste	Textile & leather	yard/wood	metals	glass	Styrofoam & sponge	C & D debris	Soils & fines
1	23.1	16.3	0.0	0.1	7.3	1.6	0.2	7.5	35.7	8.1
2	43.8	32.9	0.7	2.5	14.5	1.9	0.0	1.4	0.3	2.0
3	23.2	33.1	0.0	3.7	12.8	6.2	0.7	1.5	2.0	16.9
4	16.8	32.9	0.2	11.2	6.2	0.3	0.3	1.0	4.5	26.9
5	54.3	21.9	0.0	0.0	12.4	1.4	0.1	0.4	2.7	6.9
6	36.4	30.4	0.0	9.9	8.1	0.5	0.3	2.4	0.6	11.5
Average	32.9	27.9	0.1	4.6	10.2	2.0	0.3	2.4	7.6	12.0

Table 4-2 Physical composition of waste-B by weight basis

Sample No.	Physical Composition (% by weight)									
	paper	plastic	food waste	Textile & leather	Yard & wood	metals	glass	Styrofoam & sponge	C & D debris	Soils & fines
1	47.5	20.0	3.5	0.3	11.5	2.2	0.3	0.3	0.7	13.8
2	34.9	15.2	6.0	1.0	4.3	4.4	0.3	4.8	2.2	27.0
3	25.7	10.9	1.6	8.4	11.4	4.3	1.4	3.2	1.9	31.3
4	26.5	22.5	0.2	1.1	7.4	9.0	1.2	1.1	0.0	31.0
5	45.8	21.5	6.7	1.0	3.3	2.3	0.4	1.7	2.2	15.0
6	45.6	16.2	4.5	1.6	5.9	2.2	0.3	0.9	0.0	22.9
7	51.3	11.8	1.5	1.9	3.5	4.8	0.3	1.8	0.0	23.2
8	43.2	16.6	2.7	0.8	4.6	3.3	0.0	0.3	0.2	28.4
9	37.8	28.2	4.8	3.9	0.8	0.6	0.0	9.8	0.0	14.0
10	55.6	14.1	4.9	1.1	11.4	0.8	0.7	7.0	0.0	4.4
Average	41.4	17.7	3.6	2.1	6.4	3.4	0.5	3.1	0.7	21.1

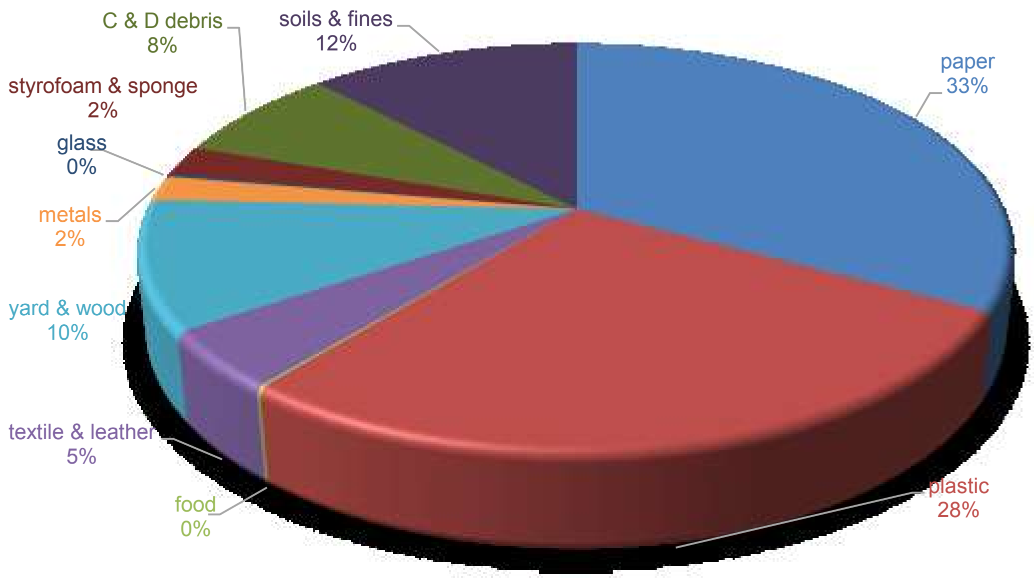


Figure 4-1 Average composition of waste-A by weight

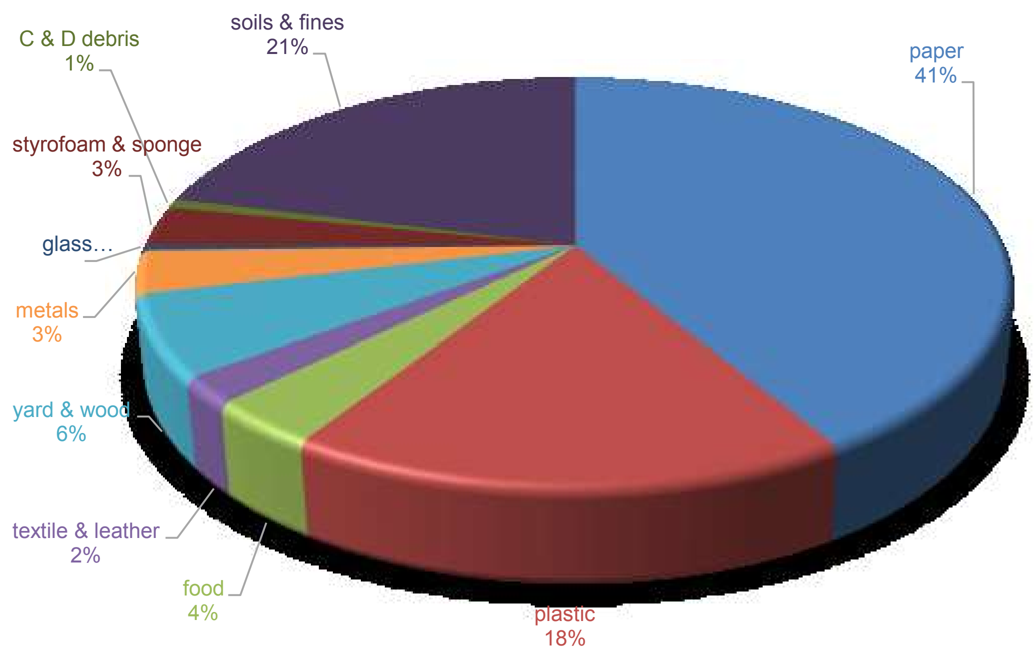


Figure 4-2 Average composition of waste-B by weight

Table 4-3 Degradable and nondegradable portions of waste-A

Sample No.	Physical Composition by degradability on weight (%)	
	Degradable	Non-Degradable
1	30.5	69.3
2	61.6	38.4
3	39.7	59.7
4	34.5	65.5
5	66.7	33.2
6	54.4	45.3
Average	47.9	51.9

Table 4-4 Degradable and nondegradable portions of waste-B

Sample No.	Physical Composition by degradability on weight (%)	
	Degradable	Non-Degradable
1	62.8	37.0
2	46.1	53.6
3	47.1	51.5
4	35.2	63.6
5	56.9	42.8
6	57.5	42.2
7	58.1	41.6
8	51.2	48.7
9	47.3	52.7
10	73.0	26.3
Average	53.5	46.0

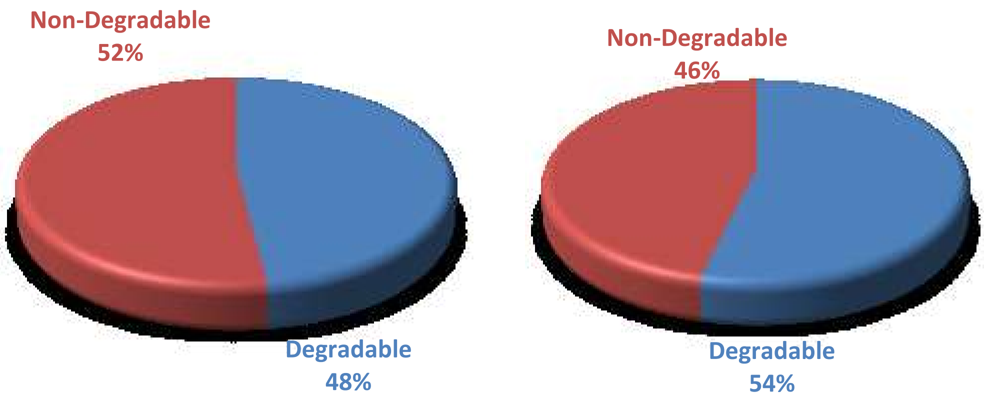


Figure 4-3 Average degradable and nondegradable amounts in waste ’A’ & waste ‘B’

The US Environmental Protection Agency has provided the national average MSW composition for year 1960 to 2012 (US-EPA, 2014) which are describes in previous chapter. The most recent data of year 2012 along with the composition of waste in the current study are summarized in Table 4-5.

The data in the Table 4-5 indicates that the amount of 'paper' in waste-A and waste-B in the current study was around 5% and 14% higher than that of national average. Similarly, the amount of 'plastics' in waste-A and waste-B was also 15.2% and 5% higher than that of and national average. 'Food' waste is an important component of MSW. The national average of the food waste is higher than that of current study. Food waste is the major contributor for the biodegradation and it degrades very quickly. The amount of 'rubber, leather, and textile' and 'wood and yard trimming' in the current study are much lower than those of the national average. The amounts of 'metal' and 'glass' in the current study were almost negligible as compared to the others. In summary, the amounts of non-degradable waste in the current study are 48% and 54% in waste-A and waste-B, respectively. The results indicate that high amount of non-degradable waste found in the current study.

Table 4-5 Comparison of physical composition of MSW

Components	USEPA 2012,%	Current study	
		Waste-A,%	Waste-B,%
Paper	27.4	32.9	41.4
Plastic	12.7	27.9	17.7
Food	14.5	0.1	3.6
Textile, leather and rubber	8.7	4.6	2.1
Wood	6.3	10.2	6.4
Yard trimmings	13.5		
Glass	4.6	0.3	0.5
Styrofoam & sponge	NA	2.4	3.1
C&D		7.6	0.7
Other fines & soil	3.4	12	21.1
Metal	8.9	2	3.4
Total	100	100	100
% Degradable	70.4	48	54
% Non-degradable	29.6	52	46

4.3 Moisture content

This test was performed to determine the water (moisture) content of solid waste. The water content is the ratio, expressed as a percentage, of the mass of “pore” or “free” water in a given mass of soil to the mass of the dry solid waste or wet solid waste. The moisture contents were determined on the dry as well as wet basis.

The moisture content of the fresh waste was determined at the time of physical composition tests. The moisture content of waste samples before putting into the simulated bioreactor and at the end of the degradation of the three reactor samples by drying approximately 2 lbs. of sample. The moisture contents of both wastes for all bags were described in Table 4-6 and Table 4-7 for waste-A and waste-B, respectively.

Table 4-6 Moisture content of fresh MSW in waste-A samples

Sample No.	Moisture Content (%)	
	Wet wt. basis	Dry wt. basis
1	33.3	50.0

Table 4-6 *continued*

2	30.1	43.1
3	26.7	36.5
4	22.6	29.2
5	26.2	35.5
6	27.1	37.2
Average	27.7	38.6

Table 4-7 Moisture content of fresh MSW in waste-B samples

Sample No.	Moisture Content (%)	
	Wet wt. basis	Dry wt. basis
1	32.9	49.0
2	28.6	40.1
3	29.8	40.4
4	27.2	37.4
5	31.9	46.9
6	24.1	31.6
7	31.9	46.9
8	24.1	31.7
9	29.7	42.7
10	30.7	44.3
Average	29.1	41.1

4.4 Maximum dry unit weight/density

Standard Proctor compaction tests conducted on shredded waste-A resulted in a maximum dry density of 376 kg/m^3 at 62% optimum moisture content (see Figure 4-4). Similarly Standard Proctor compaction tests conducted on shredded waste-B resulted in a maximum dry density of 410 kg/m^3 at 58% optimum moisture content (see Figure 4-4) Hettiarachchi (2005) reported a maximum dry density of 525 kg/m^3 at 62% optimum moisture content for a MSW sample generated in the laboratory. The author used maximum particle size 12.5 mm in his study. Reddy et al (2009 b) reported a maximum dry density of 420 kg/m^3 was observed at 70% optimum moisture content and they limited the maximum particle size to 40 mm in their study. The mix proportion for this lab-prepared MSW was selected to simulate the average MSW composition as in the Table 4-1 and Table 4-2. The difference in the maximum particle sizes and average specific gravity are the reasons responsible for the difference in maximum dry density values reported (Reddy et al., 2009 b). Another major reason could be the difference in the composition of waste samples and types of individual components. There could be several variations on the paper, plastic, metal, food, textiles, and soil. Basically, the difference in specific gravity of the materials can result the variation on density. There were so many variation on the specific gravity within the same components category such as soft paper, hard paper, soft plastic and hard plastic, aluminum and iron metal, varities of wood and textile. The variation on specific gravity of the components leads to the variation on density of the combined materials.

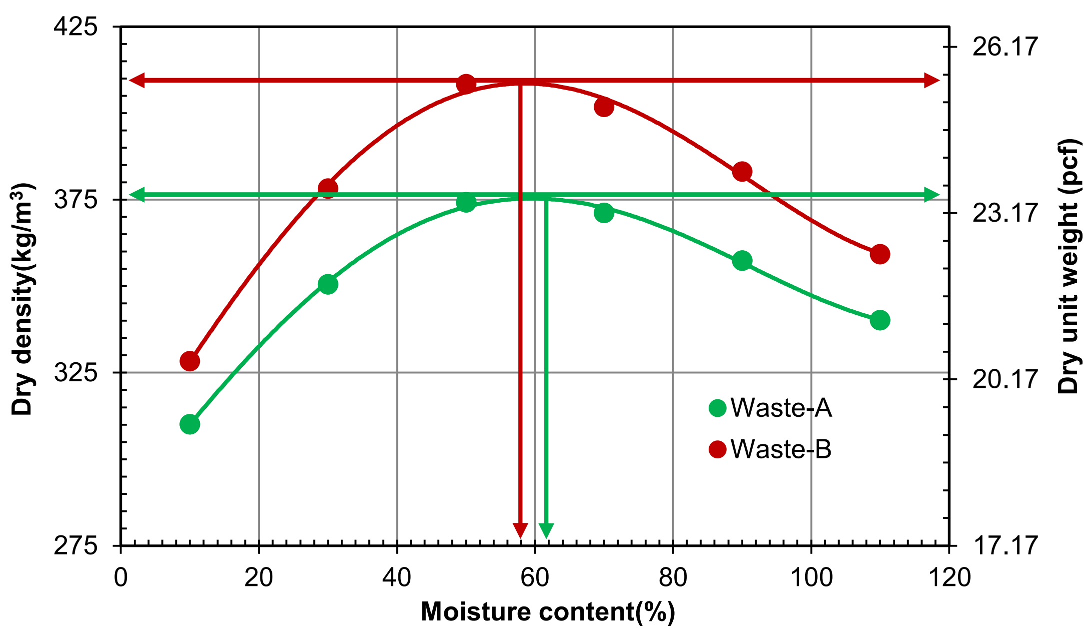


Figure 4-4 Variation of dry density of fresh shredded MSW with moisture content (dry basis)

4.5 Variation of hydraulic properties with compaction

In this section, the variation of hydraulic conductivity, porosity and moisture holding capacity were discussed. Various tests were being performed on 6.35 cm, 15.24 cm and 25.4 cm diameter permeameter, respectively. The detail of the experimental program is explained in chapter 3 (see Table 3-2).

4.5.1 Saturated hydraulic conductivity of municipal solid waste at various density

Total 51 constant head permeability tests were carried out on various fresh MSW such as shredded waste-A, shredded waste-B and un-shredded waste-B. The tests were carried out in three different size permeameters for the shredded 'waste-A' and shredded 'waste-B', respectively while tests were carried out in large size (25.4 cm) diameter permeameter for un-shredded waste-B.

Total 22 permeability tests were conducted on the waste-A sample. All samples had the exactly similar composition. The shredded waste-A had permeability range from 2.76E-02 cm/s to 2.60E-06 cm/s at the dry density between 347.7 kg/m^3 and 714.3 kg/m^3 while using a small-scale 6.35cm diameter rigid-wall permeameter. Similarly, the similar waste had permeability range from 5.38E-03 cm/s to 3.09E-06 cm/s at the dry density between 324.2 kg/m^3 and 733.1 kg/m^3 while using a medium-scale 15.35 cm diameter rigid-wall permeameter. The large 25.4 cm diameter permeameter resulted the permeability 4.96E-03 cm/s to 9.87E-06 cm/s at the dry density between 332.1kg/m^3 and 722.1 kg/m^3. The dry unit weight of these samples were varied in a big range in order to get the variation of permeability with small increment of density. The result of permeability for all waste-A samples are shown in Table 4-8 and Figure 4-5.

Similarly 22 permeability tests were conducted on the waste-B samples. The shredded waste-B had permeability range from 1.29E-02 cm/s to 1.64E-06 cm/s at the dry density between 319.5 kg/m^3 and 730.2 kg/m^3 while using small size permeameter.

Similarly, the waste-B had permeability range from 3.69E-03 cm/s to 2.22E-06 cm/s at dry density between 342.0 kg/m^3 and 734.0 kg/m^3 while using a medium-size permeameter. The large permeameter resulted permeability of 4.28E-03 cm/s to 9.54E-06 cm/s at the dry density from 322.9 to 723.7 kg/m^3. The result of permeability for all waste-B samples are shown in Table 4-9 and Figure 4-6.

Similarly, permeability tests were conducted on unshredded waste-B on seven samples at same composition using the large permeameter. The unshredded waste-B resulted permeability from 4.45E-03 cm/s to 9.65E-07 cm/s for the dry density 315.8 kg/m^3 and 720.9 kg/m^3. The hydraulic conductivity obtained from large-scale rigid wall permeameter tests are summarized in Table 4-10 and Figure 4-7.

The results clearly demonstrated that the hydraulic conductivity of MSW significantly decreased by increasing density. This was mainly attributed to the increase in density leading to low void ratio. These results showed there is correlation between the dry density and hydraulic conductivity of waste. The general trend is that the hydraulic conductivity decreases with increasing dry density for fresh MSW. The results are in agreement with the data published by Blieker et al., (1993). Reddy et al., (2009 a) also reported the hydraulic conductivity obtained from different permeameter tests decreased with the increase in dry unit weight for both fresh and landfilled waste. The higher confinement increases the density; therefore, hydraulic conductivity decreases with the increase in the confinement pressure. Zero confinement simulates fresh MSW located near the top surface of a landfill. It should be noted that the tests were conducted using saturated fresh MSW; therefore, the hydraulic conductivity values represent the saturated hydraulic conductivity of fresh MSW. If the MSW is unsaturated, then unsaturated hydraulic properties should be determined.

Table 4-8 Permeability of shredded waste-A at various density

@ small permeameter		@medium permeameter		@ large permeameter	
Density (kg/m^3)	Permeability (cm/s)	Density (kg/m3)	Permeability (cm/s)	Density (kg/m^3)	Permeability (cm/s)
347.7	2.76E-02	324.2	5.38E-03	332.1	4.96E-03
422.9	7.91E-03	399.1	2.61E-03	403.4	2.80E-03
488.7	2.16E-03	472.4	1.32E-03	468.4	1.48E-03
535.7	7.19E-04	537.5	5.05E-04	533.7	5.57E-04
592.1	1.14E-04	602.7	9.11E-05	592.5	1.70E-04
629.7	3.66E-05	667.8	1.35E-05	662.1	4.03E-05
667.3	1.06E-05	733.8	3.09E-06	722.1	9.87E-06
714.3	2.60E-06				

Table 4-9 Permeability of shredded waste-B at various density

@ small permeameter		@medium permeameter		@ large permeameter	
Density (kg/m^3)	Permeability (cm/s)	Density (kg/m3)	Permeability (cm/s)	Density (kg/m^3)	Permeability (cm/s)
319.5	1.29E-02	342.1	3.69E-03	322.9	4.28E-03
385.3	4.85E-03	407.2	1.84E-03	390.0	2.51E-03
451.1	1.74E-03	472.4	9.62E-04	441.1	1.41E-03
516.9	3.71E-04	537.5	2.99E-04	496.9	6.75E-04
578.0	9.55E-05	602.7	6.38E-05	569.7	2.19E-04
629.7	2.43E-05	668.0	1.11E-05	640.5	5.44E-05
681.4	6.25E-06	734.0	2.22E-06	723.7	9.54E-06
730.2	1.64E-06				

Table 4-10 Permeability of un-shredded waste-B at various density

@ 25.4cm diameter permeameter	
Density (kg/m^3)	Permeability(cm/s)
315.8	4.45E-03
385.1	2.32E-03
448.6	1.02E-03
512.6	3.78E-04
576.7	6.31E-05
642.3	9.97E-06
720.9	9.65E-07

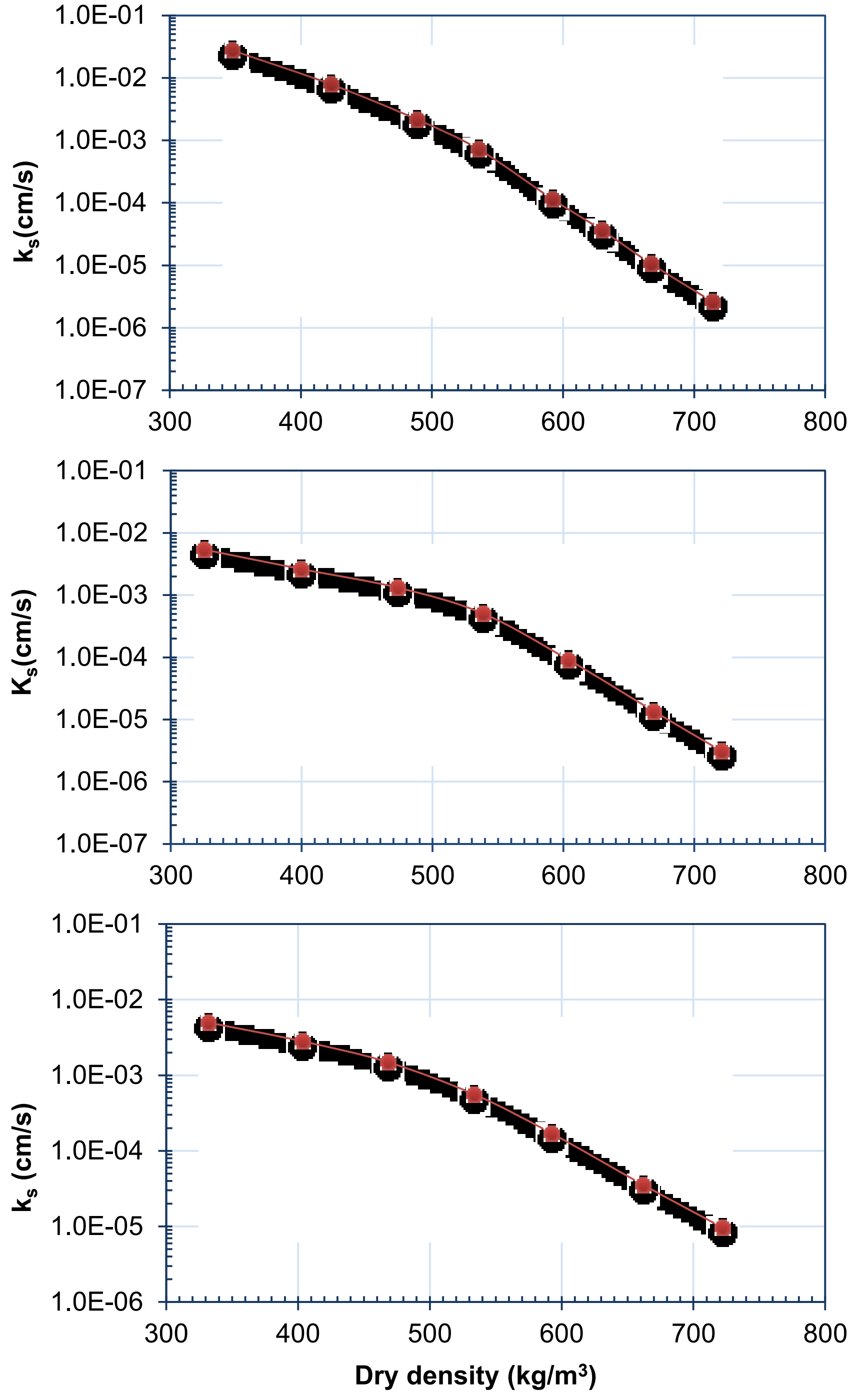


Figure 4-5 Permeability vs. density for shredded waste-A measured from 6.35 cm, 15.25 cm and 25.4 cm diameter permeameter

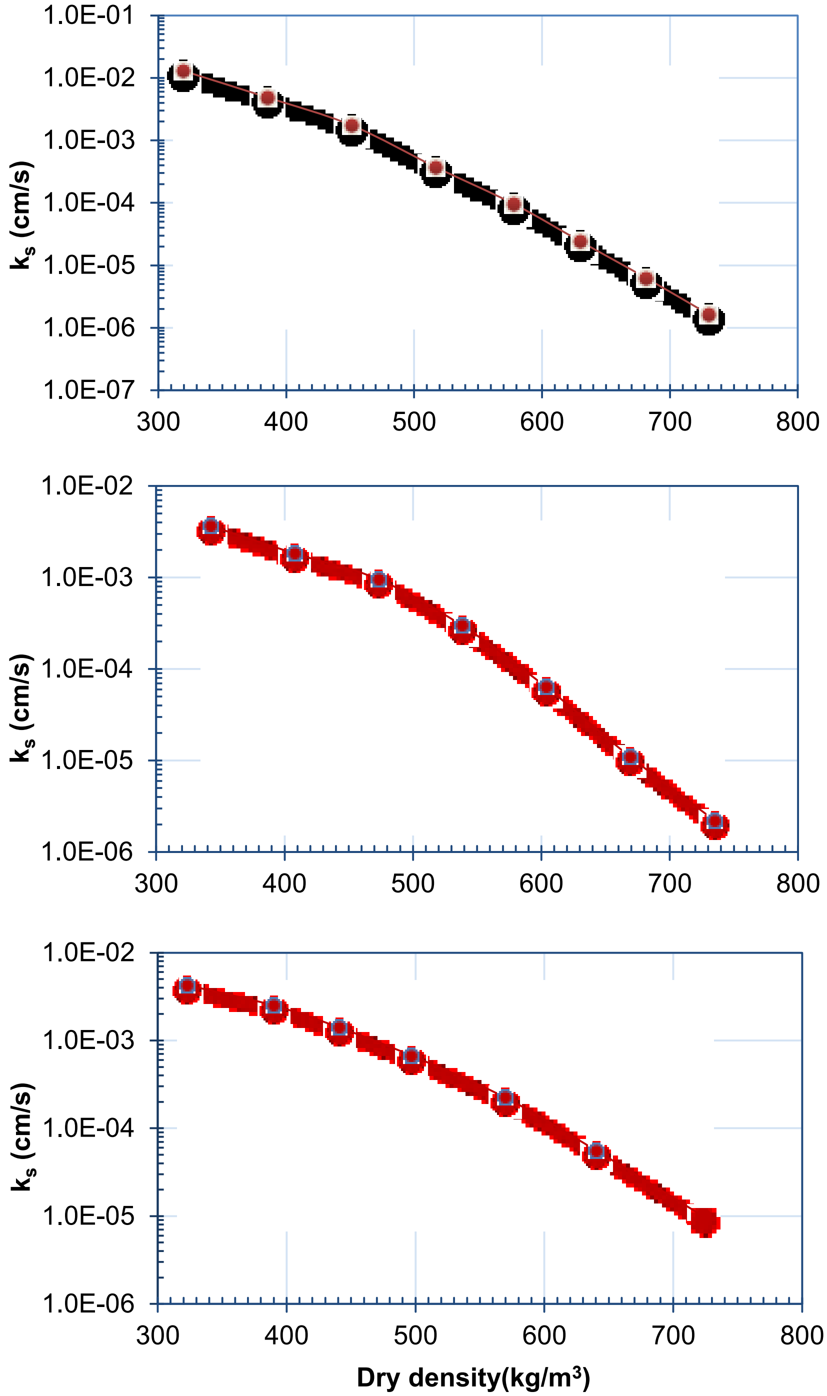


Figure 4-6 Permeability vs. density for shredded waste-B measured from 6.35 cm, 15.25 cm and 25.4 cm diameter permeameter

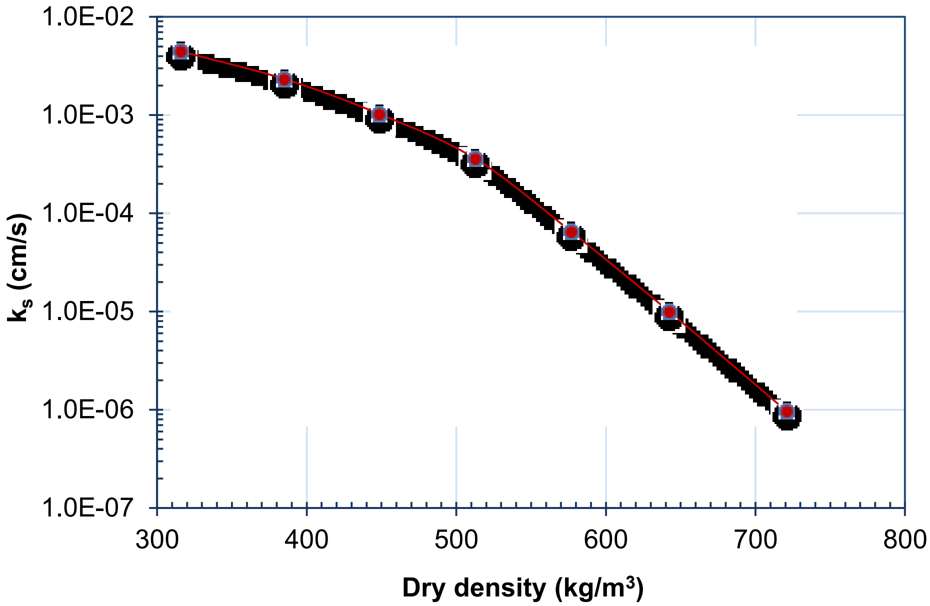


Figure 4-7 Permeability vs. density for unshredded waste-B from large diameter permeameter

4.5.2 Variation of hydraulic conductivity of waste with time

The hydraulic conductivities for waste-A were measured using constant head permeability tests. The variation in hydraulic conductivity of waste-A was recorded daily for 116 days and the results are shown in Figure 4-8. In this tests, hydraulic conductivity decreased over time for all compacted samples. Besides this, the decrease in hydraulic conductivity was more from lower to higher compacted waste samples. This may be due to filling of pore space with gas and having insufficient pore spaces available for flow of liquid. As the gas produced, the spaces were occupied with the gas and the hydraulic conductivity decreased significantly. Gas bubbles were observed in the pipe due to degradation of the waste which decreased the flow. It was important to remove gas produced in the cell otherwise accumulated gas could totally block the path of flow. The accumulated gas was released in two ways explained in previous chapter. In the first process water was continuously flowing from the reservoir tank through the upper part of the sample. Gas vent was opened with continuous flow of the water cannot remove the entire air but removed only accumulated gas on the pipe and top part of the waste sample. When the permeability tests were performed by removing the air in this way, the hydraulic conductivity was increased in a small amount. Similarly in the second process, the water tank connected to the upper part of the sample was disconnected and connected to the bottom part of the sample and water was applying into the waste sample to remove the entire accumulated air. This process can remove the entire air accumulated sample which significantly increases the hydraulic conductivity. The procedure of gas removal from the sample had a significant effect on the hydraulic conductivity which is shown in Figure 4-8**Error! Reference source not found.**. When gas was accumulated in the sample, it lowered the hydraulic conductivity in higher amount. When the permeability tests were performed by removing the entrapped by applying water from bottom of sample the flow was increased significantly. The releasing of gas from sample was frequent for highly compacted waste as compared to loosely compacted. The permeability of the waste sample was increased more when the gas was released by second process as compared to first process. This is due to filling of more voids with water which created the saturated condition. This can be concluded as unsaturation effect on permeability though it was difficult to measure degree of unsaturation. As the saturation of the sample was higher, the sample had higher permeability. When more gas was released through the second process, there was possibility of getting higher saturation as compared to first process. The frequency of removing gas was less in the lowest compacted waste sample. Powrie et al., (2005) indicated hydraulic conductivity, total and drainable porosity decreased with increase in density. The lowest density 443.2 kg/m^3 had higher total and drainable porosity as compared to other two more densified samples which was observed in previous tests results. Density is one of the most important parameter of flow along with degree of saturation. Besides this, the void space was higher in less compacted waste than highly densified samples. The low compacted waste had more interconnecting void space available for the flow and it can also store high amount of the generated gas. If gas was produced due to degradation, the lesser compacted waste sample had enough space to store generated gas. This might be the primary reason the generated gas did not frequently come to pipe in lesser compacted waste sample.

The permeability varies from 4.14×10^{-3} cm/sec to 1.33×10^{-5} cm/sec which has maximum to minimum ratio of 312 in first lowest densified sample. The lowering of permeability of the waste with time was primarily attributed due to entrapped gas inside the sample. Other factors such as potential clogging of geocomposite at the bottom of the waste sample may also have contributed to the lowering the hydraulic conductivity. The reasons of having high ratio of 312 was less frequently releasing of the gas from the sample. Similarly high degradation and continuous generation of gas might be another reason for continuous reduction of the hydraulic conductivity. Similarly the frequency of gas released in medium compacted sample of 569.8 kg/m^3 was slightly higher than the lowest densified sample. The coefficients of permeability vary from 1.87×10^{-3} cm/sec to 6.73×10^{-6} cm/sec for this medium compacted waste sample which has maximum to minimum permeability ratio 278.

Besides these samples, the frequency of removal of gas was significantly higher for the highest compacted waste sample which had dry density 696.5 kg/m^3. As the composition was same, the highest densified sample had lowest total and drainable porosity than other two compacted waste samples; so that highly compacted waste had less void space available for flow of water. If there is gas generation inside the waste sample, the compacted waste might not have enough space to store the generated gas inside the permeameter. This might be the reason for gas being observed frequently in upper pipe. The coefficient of hydraulic conductivity decreased significantly within a few days if gas was not removed from waste. If the gas was released through first process (as described previous chapter), the hydraulic conductivity increased in smaller amount but if the gas was released through second process the hydraulic conductivity increased significantly higher than previous process. The coefficients of permeability vary from 5.10××10^{-5} cm/sec to 5.90×10^{-7} cm/sec for the highly compacted sample which has maximum to minimum permeability ratio 86.4. One of the reasons of having small ratio was more frequently removing the gas from the sample.

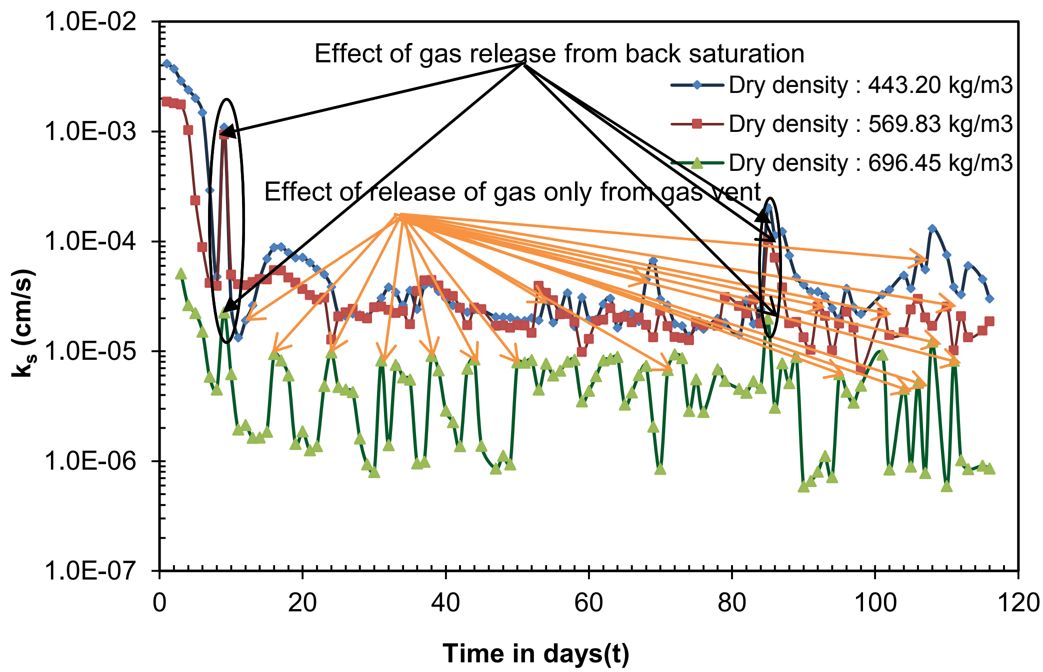


Figure 4-8 Variation of permeability for variously compacted waste-A with time

4.5.3 Porosity of municipal solid waste

Waste samples were saturated by applying tap water from the bottom of the samples for at least 24 hours in order to remove all the entrapped air. After measuring the permeability of the waste samples, the weight of the saturated samples with permeameters was measured. The weight of fully saturated waste samples were calculated by subtracting mold weight without waste samples. If gas was generated, the gas was released from the gas vent, in order to fill all the void space with water. Waste can have different types of porosity because of its nature. Generally waste can never get saturated in field condition. After exceeding the field capacity of waste, it start to drain water. In literature, researchers used different notation for different porosity. In this study, three types of porosity are used which are discussed in subsequent heading.

The water was applied till the air bubble were completely removed from waste samples. Actually saturating of waste samples was quite difficult and time consuming. It was not possible to get 100% saturation by removing all the entrapped air from the samples. While performing the tests in clear transparent permeameter, entrapped air can be observed. In order to make complete saturation, the weight of saturated waste was monitored. If the weight was remain constant, the weight was finalized as saturated weight although there was entrapped air inside the waste samples. A total 57 tests were carried out on various fresh MSW such as shredded waste-A, shredded waste-B and un-shredded waste-B on the similar samples after performing permeability tests. After measuring the weight of saturated MSW samples, the samples were allowed to drain under gravity flow. Researchers such as (Powrie et al., 1995; Powrie et al. 2005; Beaven 2000; Beaven et al., 2011) used extensively the terms total porosity and drainable porosity in their research. They have not defined the porosity at the absorptive capacity or at field capacity which is also one of the major important parameter for the bioreactor landfill. Basically in this research, porosity was estimated at fully saturated conditions, at total absorptive capacity after complete drainage. The tests were carried out in three different size permeameters as explained before.

4.5.3.1 Total porosity of MSW

Various porosities were measured on all total 25 waste-A samples after performing saturated permeability tests. The shredded waste-A had total porosity ranging from 74.3% to 40.0% for the dry density between 282.0 kg/m^3 and 714.3 kg/m^3 while using a small-size permeameter. Similarly, the waste-A had total porosity ranging from 74.6% to 39.2% for the dry density between 275.3 kg/m^3 and 733.8 kg/m^3 while using a medium-size permeameter. The large permeameter resulted in a range of total porosity of 71.0% to 46.1% for the dry density between 332.1kg/m^3 and 722.1 kg/m^3. The dry unit weight of these samples were varied in a big range at small increment in order to get the variation of porosity with the function of density. The results of total porosity for all waste-A samples are shown in Table 4-11, Table 4-12 and Table 4-13. Similarly the results of total porosity are also explained in the Figure 4-9. Similarly, various porosities were measured on all 25 waste-B samples using the same three devices after performing saturated permeability tests. The shredded waste-B had total porosity ranging from 74.4% to 38.2% for the dry density between 267.8 kg/m^3 and 730.2 kg/m^3 while using a small-size permeameter. Similarly, the waste-B has the value of total porosity ranging from 74.2% to 38.3% for the dry density between 275.3 kg/m^3 and 733.8 kg/m^3 while using a medium-size permeameter. The large device resulted in a range of total porosity of 68.8% to 45.0% for the dry density between 322.9kg/m3 and 723.7 kg/m^3. The result of total porosity for all waste-B samples are shown in Table 4-14, Table 4-15 and Table 4-16. The results are also explained in Figure 4-10. Similarly, seven unshredded waste-B samples were tested for porosity after measuring saturated permeability. The tests resulted total porosity ranging from 70.0% to 42.5% for the dry density between 315.8 kg/m^3 and 720.9 kg/m^3 while using a large permeameter. The result of total porosity for unshredded waste-B samples are shown in Table 4-17 and Figure 4-11.

The results clearly demonstrated that the total porosity of MSW can be significantly influenced by the increasing density and the results were consistent with previous studies. The increasing in density of materials, bring the particles close to each other and reduce the gap between the particles. This ultimately reduce the path for water flow. These results show there might be a correlation between the dry density and total porosity of MSW. The general trend is that, total porosity decreases with increasing dry density of fresh MSW. Several others also reported the influence of density and stress on the porosity of MSW. The higher stress increases the density; therefore, total porosity decreases with the increase in the surcharge landfill.

Table 4-11 Porosity of waste-A at various dry density using small size device

Dry unit wt.(pcf)	Dry density (kg/m3)	Total Porosity	Effective or retained porosity	Drainable porosity
17.6	281.9	74.28	38.08	36.199
21.7	347.7	71.46	40.43	31.028
26.4	422.9	68.64	46.54	22.096
30.5	488.7	64.41	51.24	13.163
33.4	535.7	60.65	50.77	9.873
37.0	592.1	56.41	47.95	8.462
39.3	629.7	52.18	46.07	6.112
41.7	667.3	48.42	44.19	4.231
44.6	714.3	39.96	37.93	2.881

Table 4-12 Porosity of waste-A at various dry density using medium size device

Dry unit wt.(pcf)	Dry density (kg/m3)	Total Porosity	Effective or retained porosity	Drainable porosity
17.2	275.3	74.58	39.01	35.57
20.2	324.2	72.30	44.10	28.19
24.9	399.1	68.26	48.70	19.56
29.5	472.4	64.35	51.39	12.96
33.6	537.5	59.43	50.22	9.21
37.6	602.7	54.84	48.85	5.99
41.7	667.9	46.54	41.93	4.60
45.8	733.8	39.16	36.26	2.90

Table 4-13 Porosity of waste-A at various dry density using large size device

Dry unit wt.(pcf)	Dry density (kg/m3)	Total Porosity	Effective or retained porosity	Drainable porosity
20.7	332.1	70.95	39.60	31.35
25.2	403.4	68.39	43.95	24.45
29.2	468.4	65.90	49.18	16.71
33.3	533.7	60.29	49.20	11.09
37.0	592.5	55.95	46.70	9.24
41.3	662.1	50.52	44.98	5.55
45.1	722.1	46.06	42.36	3.70

Table 4-14 Porosity of waste-B at various dry density using small size device

Dry unit wt.(pcf)	Dry density (kg/m3)	Total Porosity	Effective or retained porosity	Drainable porosity
16.7	267.8	74.42	38.55	35.87
19.9	319.5	72.50	43.25	29.25
24.1	385.3	70.10	46.07	24.03
28.2	451.1	67.70	51.24	16.45
32.3	516.9	61.94	51.24	10.69
36.1	578.0	54.25	46.54	7.71
39.3	629.7	48.49	43.25	5.24
42.5	681.4	42.25	38.55	3.70
45.6	730.2	38.22	35.54	2.68

Table 4-15 Porosity of waste-B at various dry density using medium size device

Dry unit wt.(pcf)	Dry density (kg/m3)	Total Porosity	Effective or retained porosity	Drainable porosity
17.2	275.3	74.35	37.46	36.89
21.4	342.1	70.27	42.93	27.34
25.4	407.2	66.44	47.83	18.61
29.5	472.4	62.27	49.87	12.41
33.6	537.5	56.07	47.75	8.33
37.6	602.7	50.60	44.32	6.28
41.7	667.9	45.05	40.65	4.41
45.8	733.8	38.36	35.10	3.26

Table 4-16 Porosity of waste-B at various dry density using large size device

Dry unit wt.(pcf)	Dry density (kg/m3)	Total Porosity	Effective or retained porosity	Drainable porosity
20.2	322.9	68.82	40.56	28.27
24.3	390.0	63.99	44.73	19.25
27.5	441.1	61.68	48.06	13.62
31.0	496.9	58.57	50.48	8.09
35.6	569.7	53.57	47.18	6.39
40.0	640.5	49.29	44.44	4.85
45.2	723.7	44.98	42.01	2.97

Table 4-17 Porosity of unshredded waste-B at various dry density using large size permeameter

Dry unit wt.(pcf)	Dry density (kg/m3)	Total Porosity	Effective or retained porosity	Drainable porosity
19.7	315.8	69.99	40.30	29.70
24.0	385.1	64.28	42.24	22.04
28.0	448.6	58.84	45.18	13.65
32.0	512.6	54.75	46.26	8.49
36.0	576.7	51.01	44.86	6.15
40.1	642.3	47.33	42.86	4.47
45.0	720.9	42.53	39.57	2.96

When all the data were plotted against dry density for all waste, it showed a correlation between total porosity and dry density of solid waste. It followed a polynomial equation with 2 degree order. The Figure 4-12 showed the relationship of total porosity with dry density. The correlation coefficient, R^2 (R-squared) of the equation was 95% for the overall total porosity data. The total porosity versus dry density equation obtained as,

$$y = -6E^{-05} \times x^2 - 0.0084 \times x + 80.822 \ldots\ldots\ldots\ldots\ldots \text{(4.1)}$$

Where y is the total porosity in percentage (%) and x is the dry density. The unit of the density is kg/m^3 while calculating total porosity. The equation (4.1) can be utilized to estimate total porosity for the similar type of solid waste within narrow limit of composition variation.

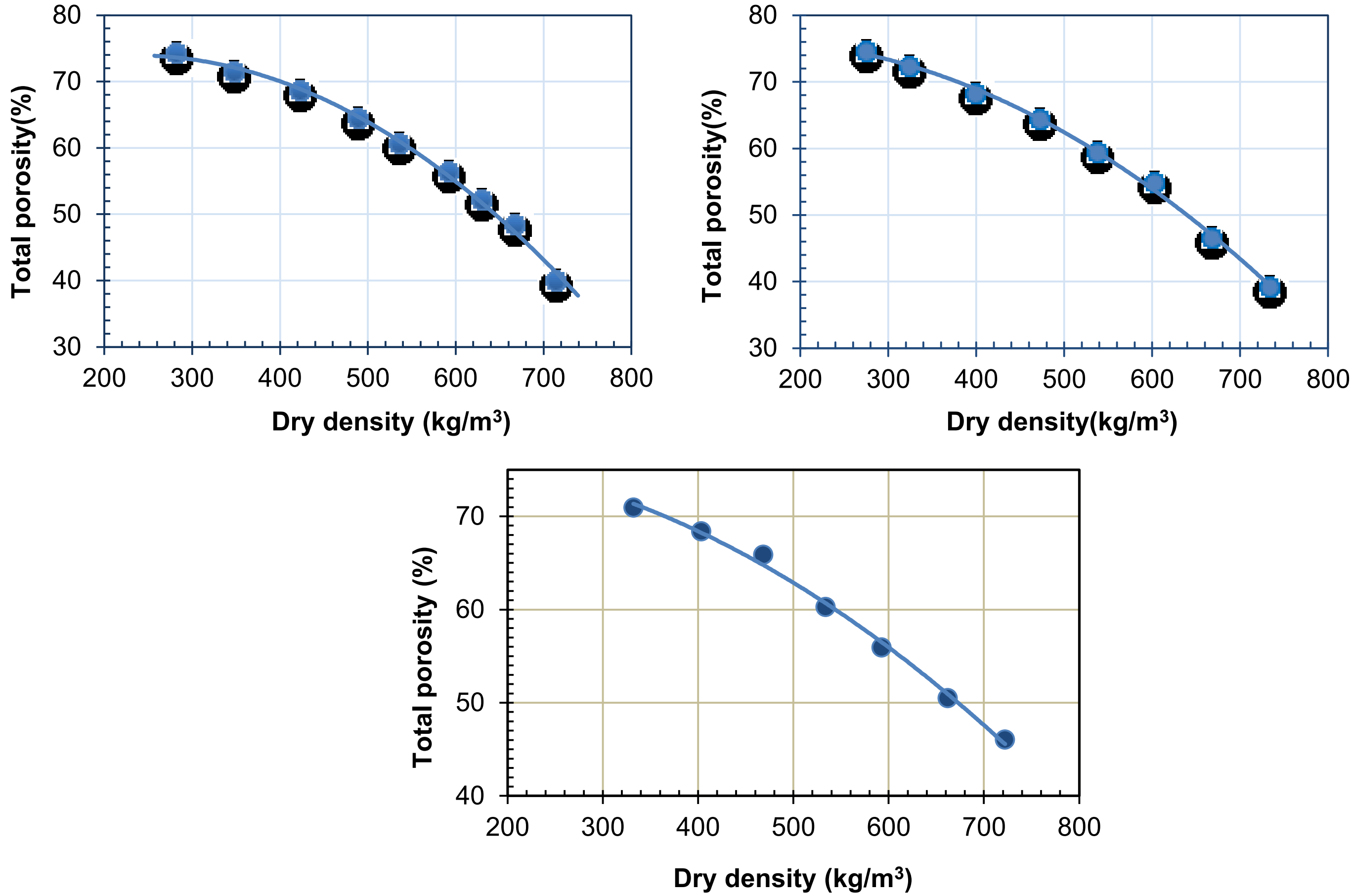


Figure 4-9 Variation of total porosity with dry density for waste-A from various size devices as small; medium and large size device, respectively

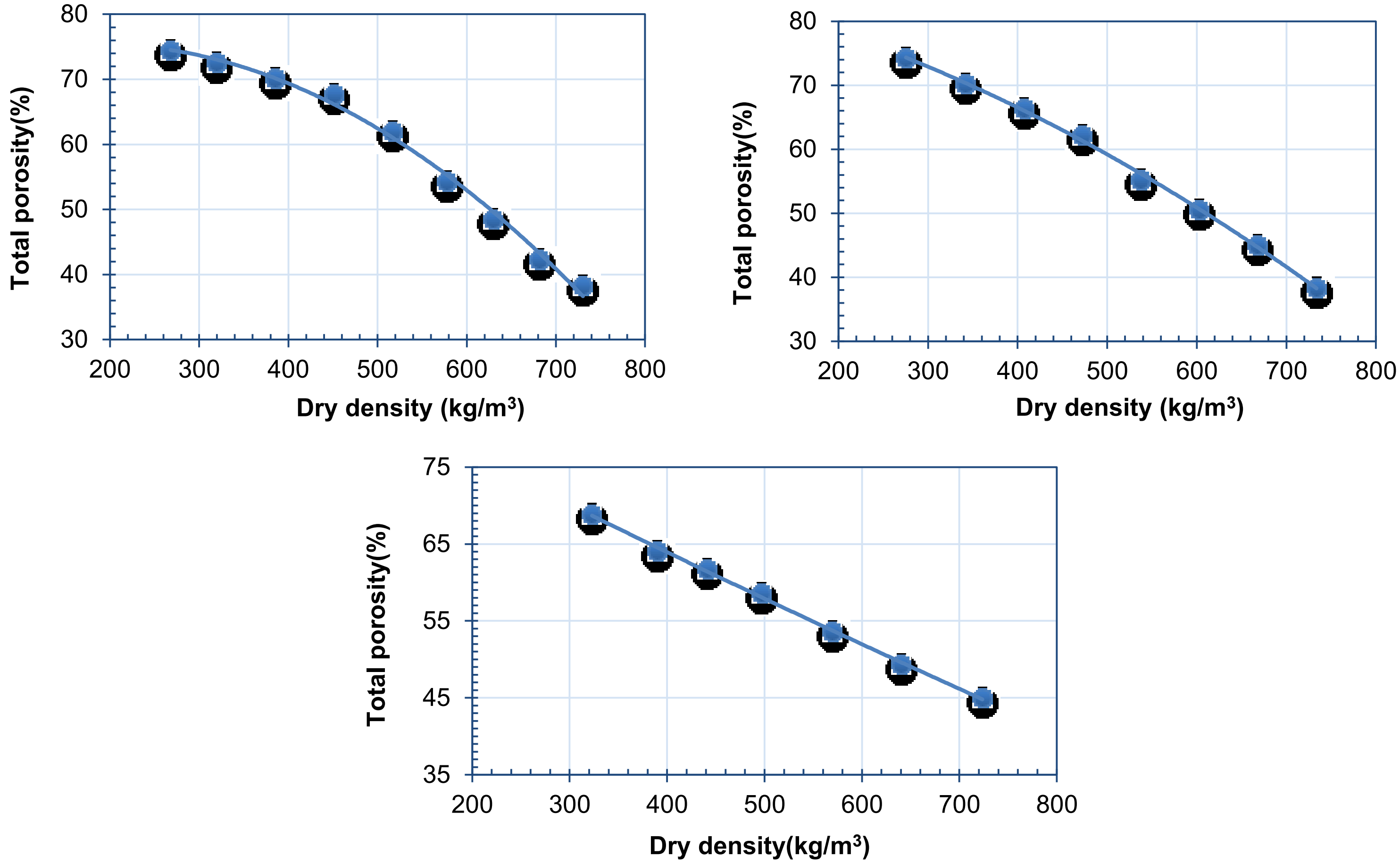


Figure 4-10 Variation of total porosity with dry density for waste-B from various size devices as small; medium and large devices, respectively

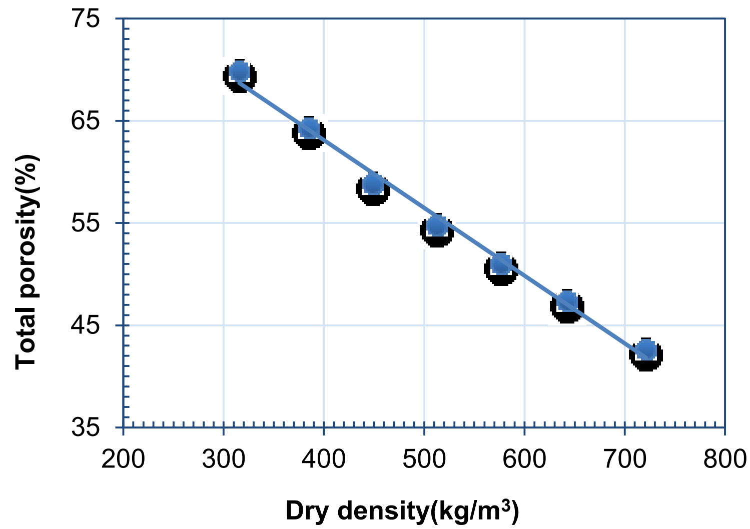


Figure 4-11 Variation of total porosity with dry density for unshredded waste

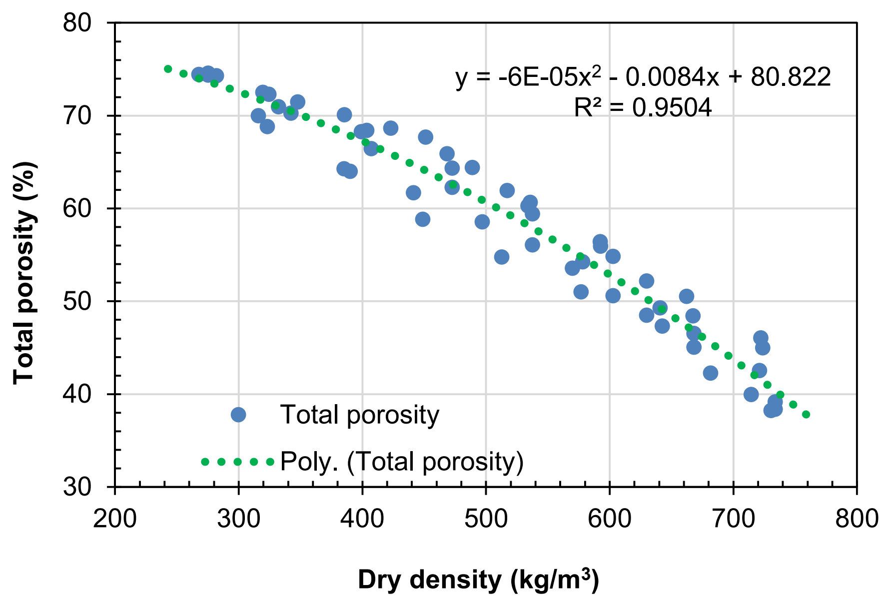


Figure 4-12 Relationship between total porosity and dry density

4.5.3.2 Retained/effective porosity of MSW

Retained porosities were calculated for all samples. Certain amount of water is always absorbed by solid waste and which cannot be removed unless applied heat and expose to external environment. It is completely different than total porosity. Total porosity decreased with increasing density whereas the retained/effective porosity was increased initially with increasing density and reached the maximum at some point. With further increasing density, porosity started to decrease. The shredded waste-A had retained porosity 38.1% at 282.0 kg/m^3 and 37.9% at 714.3 kg/m^3 for dry density through a small size permeameter. It reached the maximum of 51.2% at dry density 488.7 kg/m^3. Similarly, the waste had retained porosity of 39.0% at 275.3 kg/m^3 while 36.3% at 733.8 kg/m^3 for dry density by using a medium size permeameter. It had reached the maximum 51.4% at dry density 472.4 kg/m^3. The large permeameter resulted retained porosity of 39.6% at 332.1 kg/m^3 while 42.4% at 722.1 kg/m^3 for dry density. It reached the maximum retained porosity 49.2% at the dry density 533.7 kg/m^3. The result of retained porosity for all waste-A samples are shown in Table 4-11, Table 4-12, and in Figure 4-13. Similarly, porosities were measured on the total 25 waste-B samples. Nearly similar kind of trends were observed for waste-B. The shredded waste-B had total porosity 38.6% at 267.8 kg/m^3 and 35.5% at 730.2 kg/m^3 for dry density while using a small size diameter permeameter. It had reached the maximum 51.2% at dry density 488.7 kg/m^3 and 516.2 kg/m^3. Similarly, the waste-B has the value of retained/effective porosity 37.4% at 275.3 kg/m^3 while 34.9% at 733.8 kg/m^3 for the dry density while using a medium size diameter permeameter. It had reached the maximum retained porosity of 50.0% at dry density 472.4 kg/m^3. The large 25.4 cm diameter permeameter resulted retained porosity of 40.6% at 322.9 kg/m^3 while 42.0% at 723.7 kg/m^3 for the dry density. The waste reached the maximum retained porosity 50.5% at the dry density 496.9 kg/m^3. The result of retained porosity for all waste-B samples are shown in Table 4-14, Table 4-15, Table 4-16 and in Figure 4-14. Similarly, seven unshredded waste-B samples were utilized to measure retained porosity. The unshredded waste-B had retained porosity of 40.3 % at the dry density 315.8 kg/m^3 while 39.7% at the dry density 720.9 kg/m^3 while using a large size permeameter. The waste reached the maximum retained porosity 46.3 % at the dry density 512.6 kg/m^3. The result of total porosity for unshredded waste-B samples are shown in Table 4-17 and Figure 4-15. The results clearly demonstrated that the retained porosity can be significantly influenced by the density. Initially, the retained porosity increased with increasing dry density and reached a peak and then started to decrease with increasing dry density This was an interesting results for waste materials in terms of porosity. These results show there might be a correlation between the dry density and retained porosity of MSW.

When all the retained porosity values were plotted against dry density for all type of wastes, it showed a direct relationship between retained porosity and dry density for solid waste. Similarly the retained porosity followed a polynomial line equation with 2 degree order in dry density. The Figure 4-18 explains the relationship of retained porosity with dry density. The correlation coefficient, R^2 (R-squared) of the equation was 80.5% for the overall retained porosity data. The total porosity versus dry density equation obtained as,

$$y = -0.0002 \times x^2 + 0.2205 \times x - 6.5551 \ldots\ldots\ldots\ldots\ldots. (4.2)$$

Where y is the retained porosity in percentage (%) and x is the dry density. The unit of the density is kg/m^3 while calculating retained porosity. The formula can be used to calculate retained porosity for the similar type of solid waste

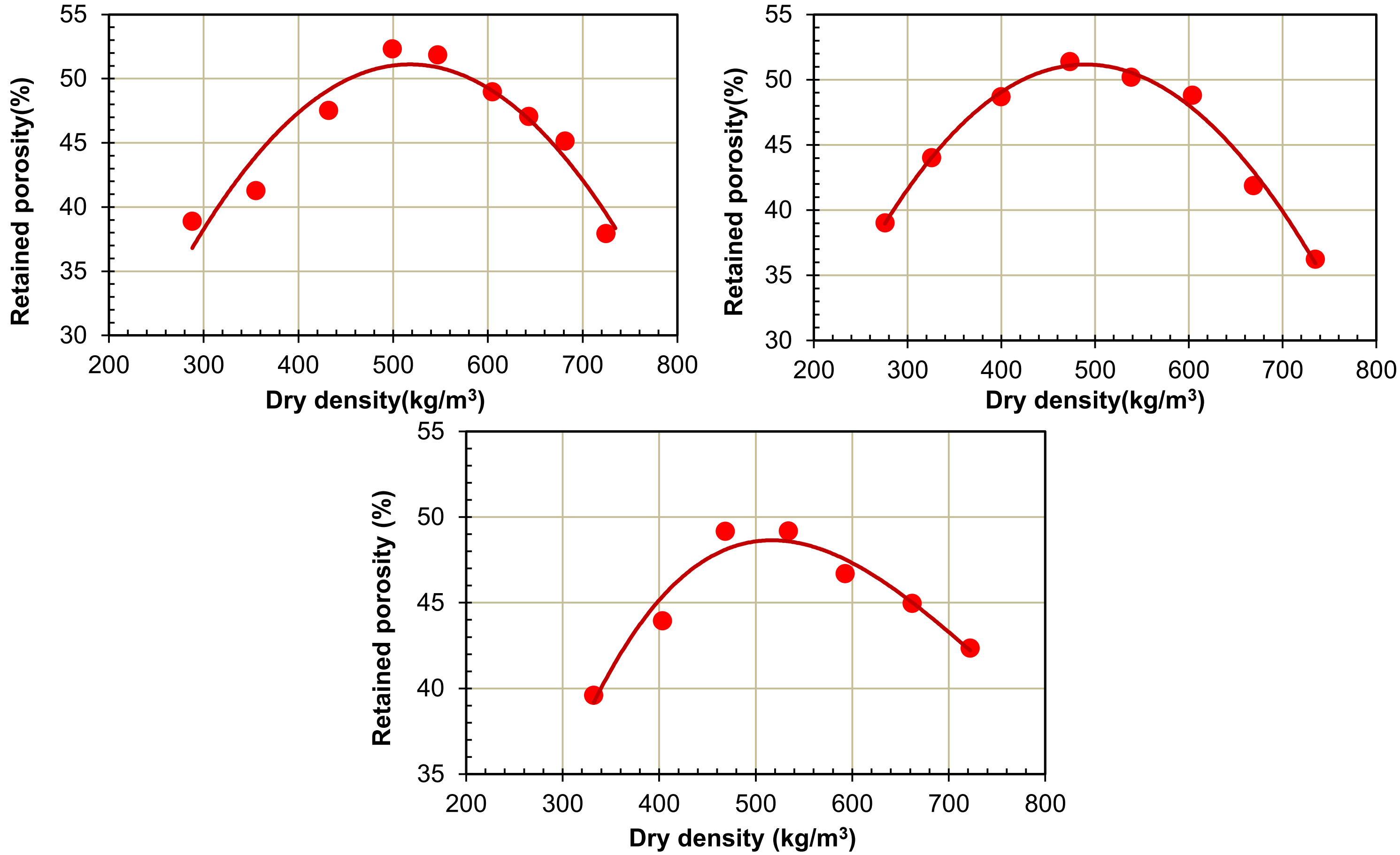


Figure 4-13 Variation of retained porosity with dry density for waste-A from various devices as small; medium and large, respectively

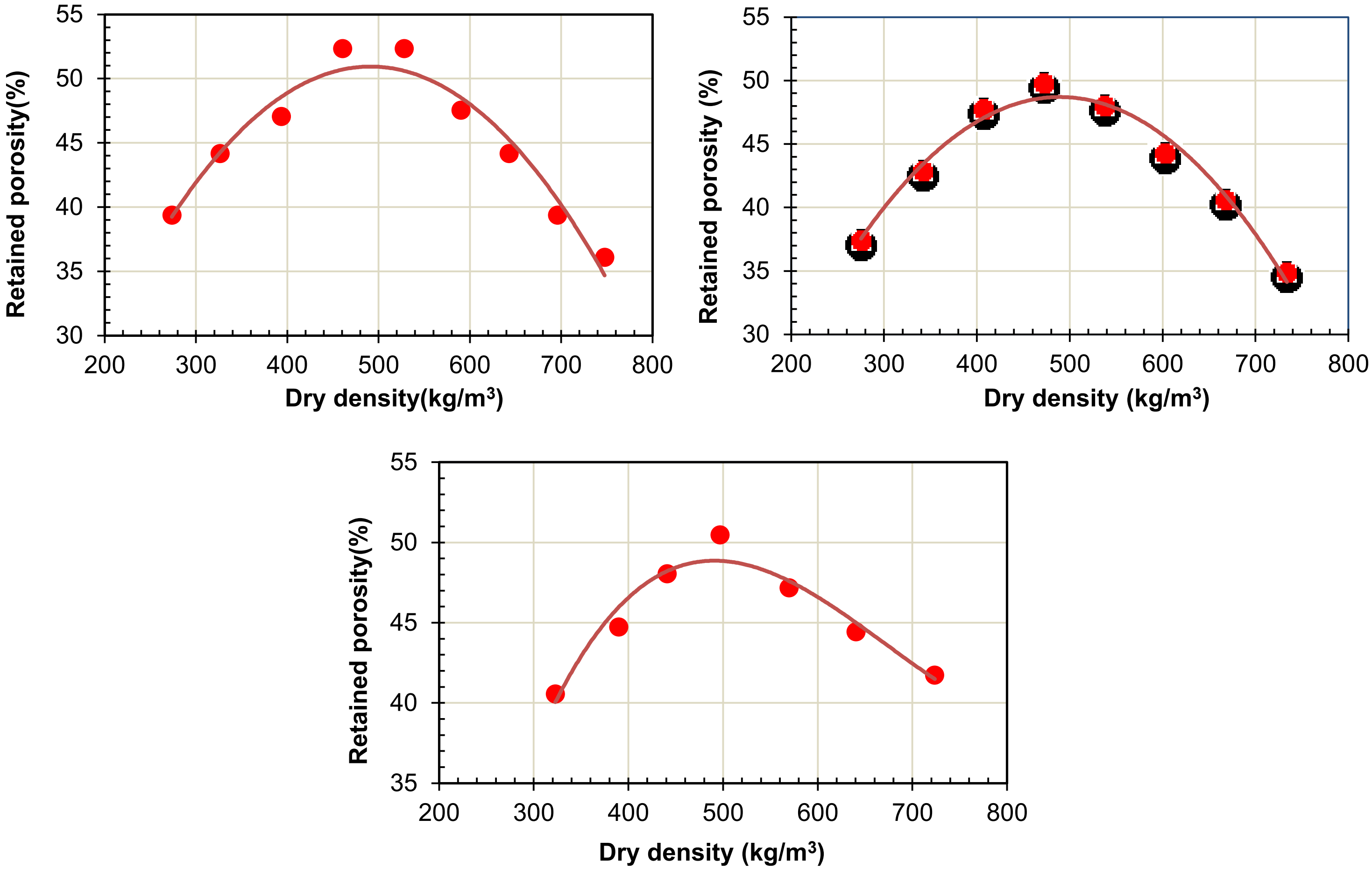


Figure 4-14 Variation of retained porosity with dry density for waste-B from various devices as small; medium and large, respectively

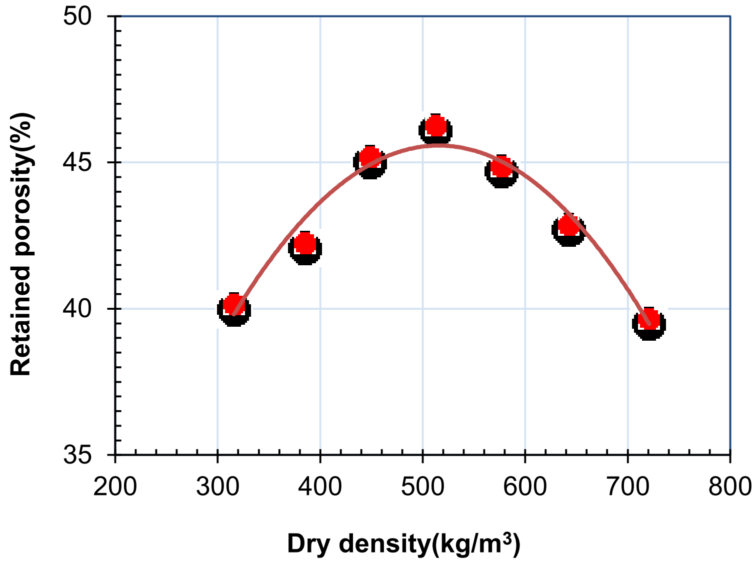


Figure 4-15 Variation of retained porosity with dry density for unshredded waste

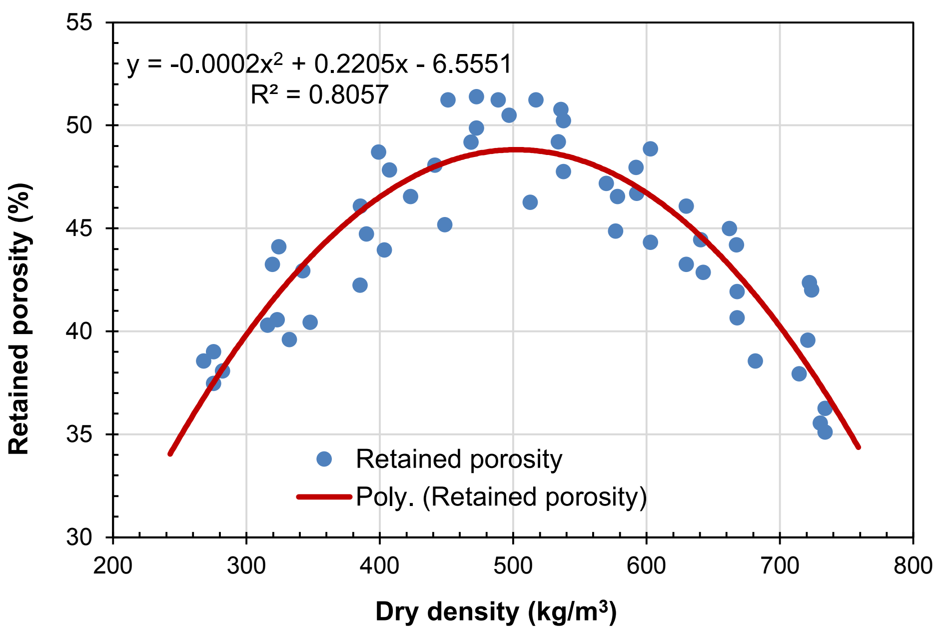


Figure 4-16 Relationship between retained porosity and dry density for waste

4.5.3.3 Drainable porosity of MSW

Similarly drainable porosities were estimated on all waste-A, waste-B and unshredded waste-B samples. Drainable porosity is the porosity different between total and retained porosity. After draining of water from saturated samples, part of the void will be occupied by air. The shredded waste-A had the drainable porosity ranging from 36.2% to 3.3% for the dry density between 281.9 kg/m^3 and 714.3 kg/m^3 while using a small-size permeameter. Similarly, the waste-A had the drainable porosity ranging from 35.6 % to 2.9% for the dry density between 275.3 kg/m^3 and 733.8 kg/m^3 while using a medium size permeameter. The large permeameter resulted drainable porosity of 31.4% to 3.7% for the dry density between 332.1kg/m^3 and 722.1kg/m^3. The results of drainable porosity for all waste-A samples are shown in Table 4-11, Table 4-12, and in Figure 4-17. Drainable porosity were also estimated for unshredded waste-B samples. The waste had drainable porosity ranging from 35.9 % to 2.9 % for the dry density between 267.8 kg/m^3 and 730.2 kg/m^3 while using a small size permeameter. Similarly, the waste-B had drainable porosity ranging from 36.8 % to 3.4 % for the dry density between 275.3 kg/m^3 and 733.8 kg/m^3 while using a medium size permeameter. The large 25.4 cm diameter permeameter resulted in a range of drainable porosity of 29.7% to 3.0 for dry density between 322.9 kg/m^3 and 723.7 kg/m^3. The result of drainable porosity for all waste-B samples are shown in Table 4-14, Table 4-15, Table 4-16 in Figure 4-18. Similarly, seven unshredded waste-B sample were utilized to measure drainable porosity using the large permeameter after measuring the weight of sample at stable condition. The unshredded waste-B had drainable porosity ranging from 69.8 % to 42.6 % for the dry density between 315.8 kg/m^3 and 720.9 kg/m^3 while using a large permeameter. The result of total porosity for unshredded waste are shown in Table 4-17 and in Figure 4-19.

The results clearly demonstrated that the drainable porosity of MSW significantly decreased by increasing density. This is a very important parameter connected with leachate generation and circulation capacity of solid waste. If the drainable porosity is low, the waste cannot generate leachate. Similarly, if the drainable porosity is high, the leachate will drain quickly so that the recirculation also becomes faster. These results clearly show there might be a correlation between the dry density and drainable porosity of MSW. The general trend is that the drainable porosity decreases with increasing dry density. Previous studies (Beaven 2000, Beaven et *al.,* 2011) also reported the influence of density and stress on the drainable porosity of MSW. The higher stress in the landfill increases the density. The density will increase with increasing depth in landfill; therefore, drainable porosity decreases with the increase in the surcharge landfill.

When all the drainable porosities were plotted against dry densities for all type of wastes, it showed a direct relationship between drainable porosity and dry. It showed a sharp decrease in drainable porosity after crossing 500 kg/m^3 dry density. The drainable porosity followed a polynomial line equation with 3 degree order. The Figure 4-20 explains the relationship of drainable porosity with dry density. The correlation coefficient, R^2 (R-squared) of the equation was 97.5 % for the overall drainable porosity which determined in the current research. The drainable porosity versus dry density equation obtained as,

$$y = -4\,E^{-8} \times x^3 + 0.0002 \times x^2 - 0.2611 \times x + 92.303 \ldots\ldots\ldots\ldots \text{(4.3)}$$

Where y is the drainable porosity in percentage (%) and x is the dry density. The unit of the density is kg/m^3 while calculating retained porosity. The equation (4.3) can be utilized to estimate drainable porosity for the similar type of solid waste within narrow range of variation on composition

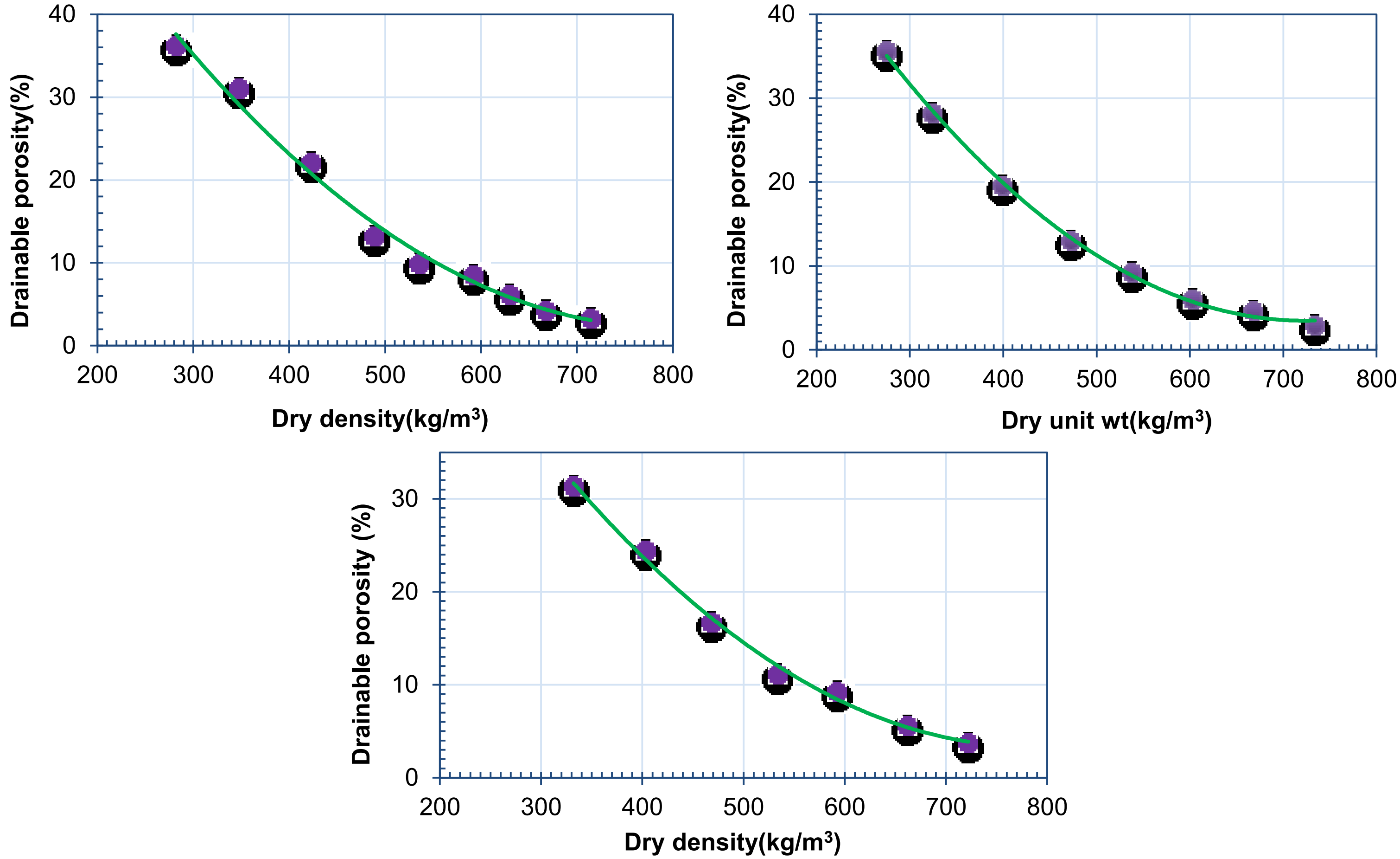


Figure 4-17 Variation of drainable porosity with dry density for shredded waste-A from small, medium and large size devices

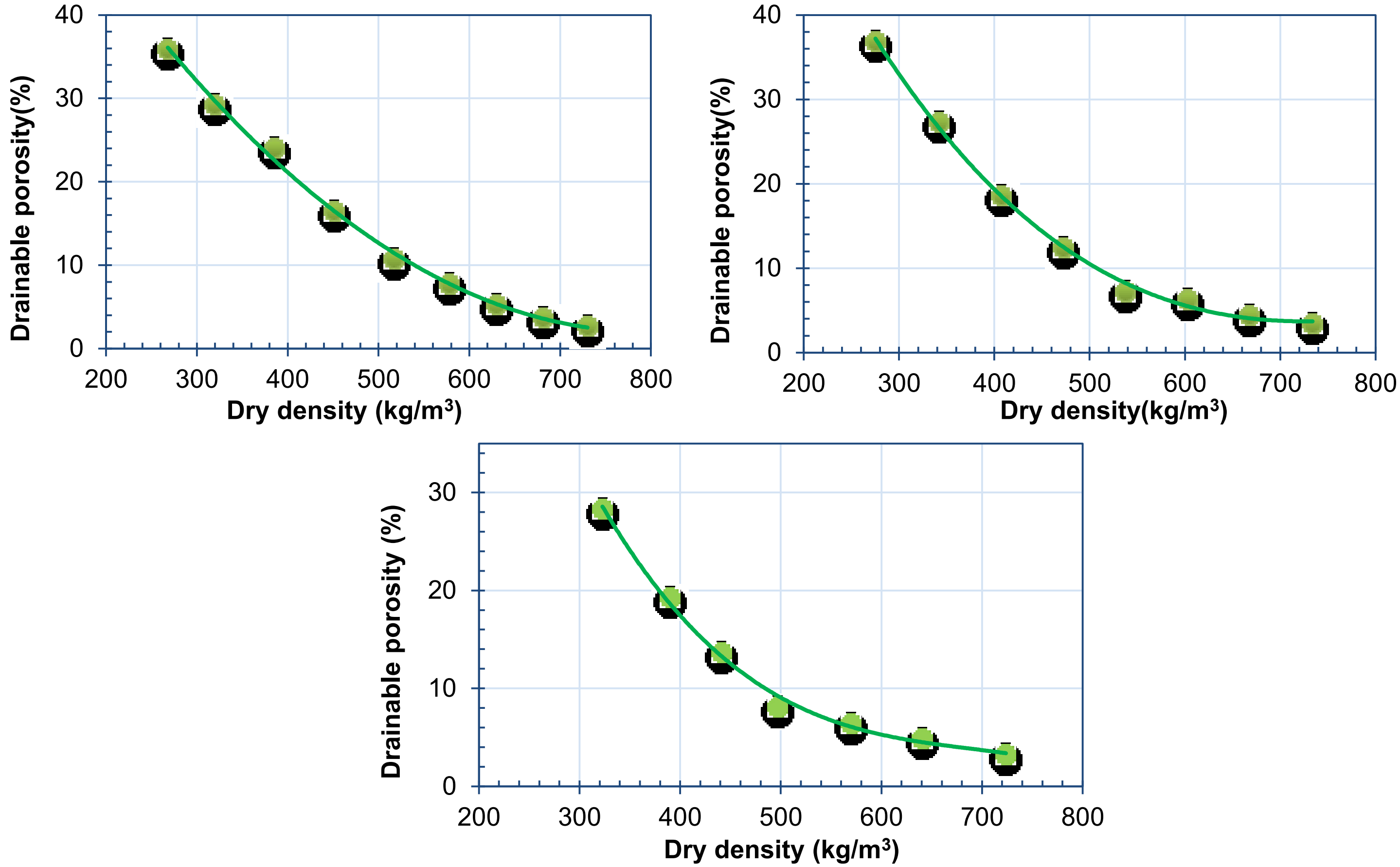


Figure 4-18 Variation of drainable porosity with dry density for shredded waste-B from small, medium and large size devices

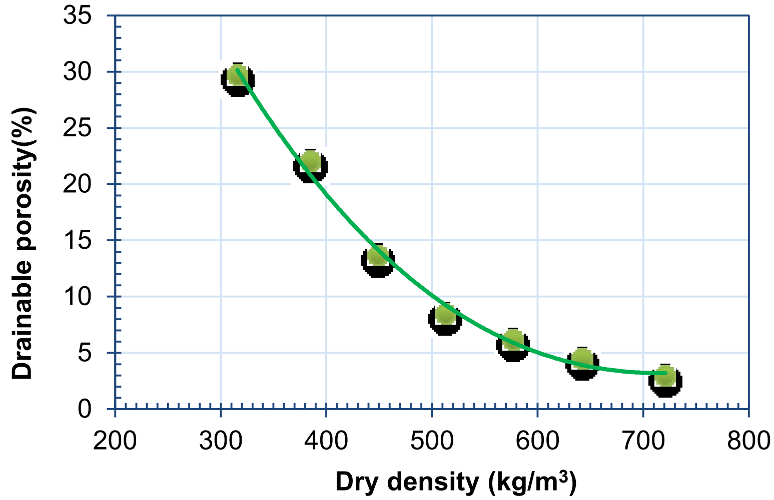


Figure 4-19 Variation of drainable porosity with dry density for unshredded waste-B obtained from large-scale device

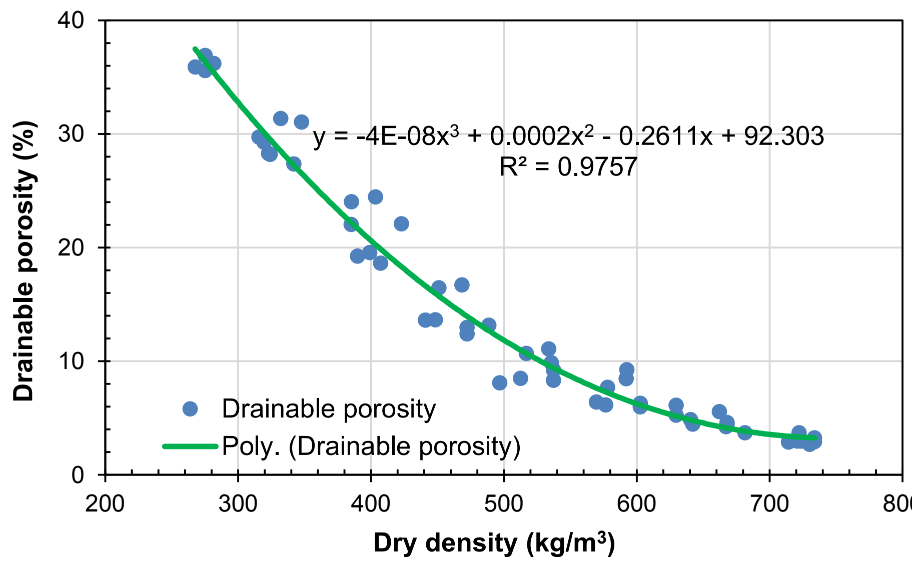


Figure 4-20 Relationship between drainable porosity and dry density for fresh MSW

4.5.4 Various moisture content of municipal solid waste

Moisture contents were determined for all samples after permeability tests. Waste samples were saturated by applying tap water from the base of the samples for at least 24 hours in order to remove all the entrapped air from the samples. In highly compacted samples, the saturation time was even more than 24 hours. Water was applied until air bubbles were observed from air vent. Complete saturation of the sample was difficult in waste. It was not possible to remove all the entrapped air from the samples. The full saturation of the samples can be achieved with incremental increase in pressure and consolidation by using the triaxial apparatus. The more saturation will achieve if fine shredded sample are used as compared to bigger particles. While using the small mold with finely shredded particles there will be less probability of entrapping air inside the samples. In order to make complete saturation, the weight of saturated waste was monitored. If the weight was remain constant, the weight was finalized as saturated weight although there was entrapped air inside the waste samples. The weight of fully saturated waste samples were calculated by subtracting mold weight without waste samples to the waste with mold. If gas was generated, the gas was released from the gas vent, in order to fill all the void space with water. Waste materials have two types of moisture content because of its nature. Generally waste can never stay in saturated condition. After exceeding the total absorptive of waste, it start to drain water and remain in a stable condition. Generally in stable condition, moisture will remain in almost constant. A number of series of tests were carried out on various fresh MSW such as shredded waste-A, shredded waste-B and un-shredded waste-B on the same samples after performing hydraulic conductivity.

After measuring the weight of saturated MSW samples, the samples were allowed to drain under gravity flow. The absorptive capacity which is one of the major important parameter for the bioreactor landfill. The moisture content of the waste should be higher than moisture retained capacity in order to enhance the degradation. It is an indicator for requirement of recirculation of water/leachate inside waste. Without exceeding the total absorptive capacity, the waste cannot generate leachate. Basically, in this research moisture content were estimated at fully saturated conditions, and at total absorptive of the waste samples.

Table 4-18 Variation of moisture content of waste 'A' measured from small device

Dry unit wt.(pcf)	Dry density (kg/m3)	Saturated moisture content (%)	Retained Moisture content (%)
17.6	281.9	263.33	135.00
21.7	347.7	205.41	116.22
26.4	422.9	162.22	110.00
30.5	488.7	131.73	104.81
33.4	535.7	113.16	94.74
37.0	592.1	95.24	80.95
39.3	629.7	82.84	73.13
41.7	667.3	72.54	66.20
44.6	714.3	55.92	51.32

Table 4-19 Variation of moisture content of waste-A measured from medium device

Dry unit wt.(pcf)	Dry density (kg/m3)	Saturated moisture content (%)	Retained Moisture content (%)
17.2	275.3	266.30	139.30
20.2	324.2	219.23	133.74
24.9	399.1	168.12	119.96
29.5	472.4	133.89	106.93
33.6	537.5	108.67	91.83
37.6	602.7	89.43	79.67
41.7	667.9	68.49	61.72
45.8	733.8	52.53	48.65

Table 4-20 Variation of moisture content of waste-A measured from large device

Dry unit wt.(pcf)	Dry density (kg/m3)	Saturated moisture content (%)	Retained Moisture content (%)
20.7	332.1	213.54	119.19
25.2	403.4	169.47	108.89
29.2	468.4	140.63	104.96
33.3	533.7	112.93	92.15
37.0	592.5	94.39	78.79
41.3	662.1	76.28	67.91
45.1	722.1	63.75	58.64

Table 4-21 Variation of moisture content of waste-B measured from small device

Dry unit wt.(pcf)	Dry density (kg/m3)	Saturated moisture content (%)	Retained Moisture content (%)
16.7	267.8	271.93	143.86
19.9	319.5	222.06	135.29
24.1	385.3	178.05	119.51
28.2	451.1	146.88	113.54
32.3	516.9	117.27	99.09
36.1	578.0	91.87	80.49
39.3	629.7	75.37	68.66
42.5	681.4	60.69	56.55
45.6	730.2	51.22	48.65

Table 4-22 Variation of moisture content of waste-B measured from medium device

Dry unit wt.(pcf)	Dry density (kg/m3)	Saturated moisture content (%)	Retained Moisture content (%)
17.2	275.3	269.53	135.80
21.4	342.1	205.00	125.24
25.4	407.2	162.80	117.20
29.5	472.4	131.55	105.34
33.6	537.5	102.58	89.24
37.6	602.7	83.78	73.38
41.7	667.9	67.32	60.73
45.8	733.8	52.16	47.50

Table 4-23 Variation of moisture content of waste-B measured from large device

Dry unit wt.(pcf)	Dry density (kg/m3)	Saturated moisture content (%)	Retained Moisture content (%)
20.2	322.9	213.07	125.56
24.3	390.0	164.00	114.65
27.5	441.1	139.76	108.90
31.0	496.9	117.82	101.55
35.6	569.7	94.00	82.78
40.0	640.5	76.92	69.35
45.2	723.7	62.13	57.66

Table 4-24 Variation of moisture contents of un-shredded fresh solid waste measured from large device

Dry unit wt.(pcf)	Dry density (kg/m3)	Saturated moisture content (%)	Retained Moisture content (%)
19.7	315.8	221.07	127.07
24.0	385.1	166.84	109.64
28.0	448.6	131.11	100.69
32.0	512.6	106.76	90.20
36.0	576.7	88.41	77.75
40.1	642.3	73.65	66.69
45.0	720.9	59.08	54.98

4.5.4.1 Saturated moisture content of municipal solid waste

Moisture contents were measured on all 25 waste-A samples. The saturated moisture contents for specific waste samples at same density were almost identical. There was small differences even in same density using the different devices. The difference was basically due to particle sizes, uneven compaction and saturation time. Generally moisture contents were slightly high for those samples in small devices with small particle sizes. The shredded waste-A has the saturated moisture content ranging from 263.3 % to 55.9 % for the dry density between 281.9 kg/m^3 and 714.3 kg/m^3 while using a small size device. Similarly, the waste-A has the value of saturated moisture content ranging from 266.3 % to 52.5 % for the dry density between 275.3 kg/m^3 and 733.8 kg/m^3 while using a medium size rigid-wall permeameter. The large permeameter resulted in a range of saturated moisture content of waste from 213.5 % to 63.8 % for the dry density between 332.1kg/m3 and 722.1 kg/m3. The results of moisture content for all waste-A samples are shown in Table 4-18, Table 4-19, Table 4-20 and in Figure 4-21.

Similarly, saturated moisture contents ranging from 271.9 % to 51.2 % for the dry density between 267.8 kg/m^3 and 730.2 kg/m^3 for waste-B while using a small size permeameter. Similarly, the waste-B had saturated moisture contents ranging from

269.5% to 52.2% for the dry density between 275.3 kg/m^3 and 733.8 kg/m^3 while using a medium size permeameter. The large permeameter resulted in a range of saturated moisture contents from 213.1 % to 62.1 % for the dry density between 322.9 kg/m^3 and 723.7 kg/m^3. The result of moisture contents for all waste-B samples are shown in Table 4-21, Table 4-22, Table 4-23 and in Figure 4-22. Similarly, saturated moisture contents were measured on the waste-B samples using the large permeameter device. The unshredded waste-B had the value of saturated moisture contents ranging from 221.1% to 59.1% for the dry density between 315.8 kg/m^3 and 720.9 kg/m^3. The result of moisture contents for unshredded waste-B samples are shown in Table 4-24 and Figure 4-23**.**

The results clearly demonstrated that the saturated moisture content can be greatly influenced by increasing density. The increasing in density of materials, bring the particles close to each other and reduce the gap between the particles. The volume of the voids in the waste was reduced so that the water content of the waste also reduced.. The general trend was that the saturated moisture content decreased with increasing in dry density of fresh MSW. Several previous studies also reported the influence of density on the moisture content of MSW.

When all the saturated moisture contents were plotted against dry densities for all of MSW samples, it followed a perfect relationship between saturated moisture content and dry density. The saturated moisture content (M.C.) followed an equation with 2 degree order. The Figure 4-29 explains the relationship of saturated moisture content (M.C.) with dry density. The correlation coefficient, R2 (R-squared) of the equation was 99 % for the overall saturated moistures which were determined in the current research. The saturated moisture content versus dry density equation can be obtained as,

$$y = 0.0008 \times x^2 - 1.212 \times x + 532.9 \ldots\ldots\ldots\ldots \text{(4.4)}$$

Where y is the saturated moisture contents in percentage (%) and x is the dry density of fresh municipal solid waste. The unit of the density is kg/m^3 in the equation (4.4) while estimating saturated moisture contents. The equation (4.4) can be utilized to estimate saturated moisture content for the similar type of solid waste within narrow range of variation on composition of MSW. The waste-A and waste-B consisted mainly paper and plastic as dominant components. If the waste contains large amount of textile and wood/yard, the equation might not be useful because those components amount were low in the waste used in the current study.

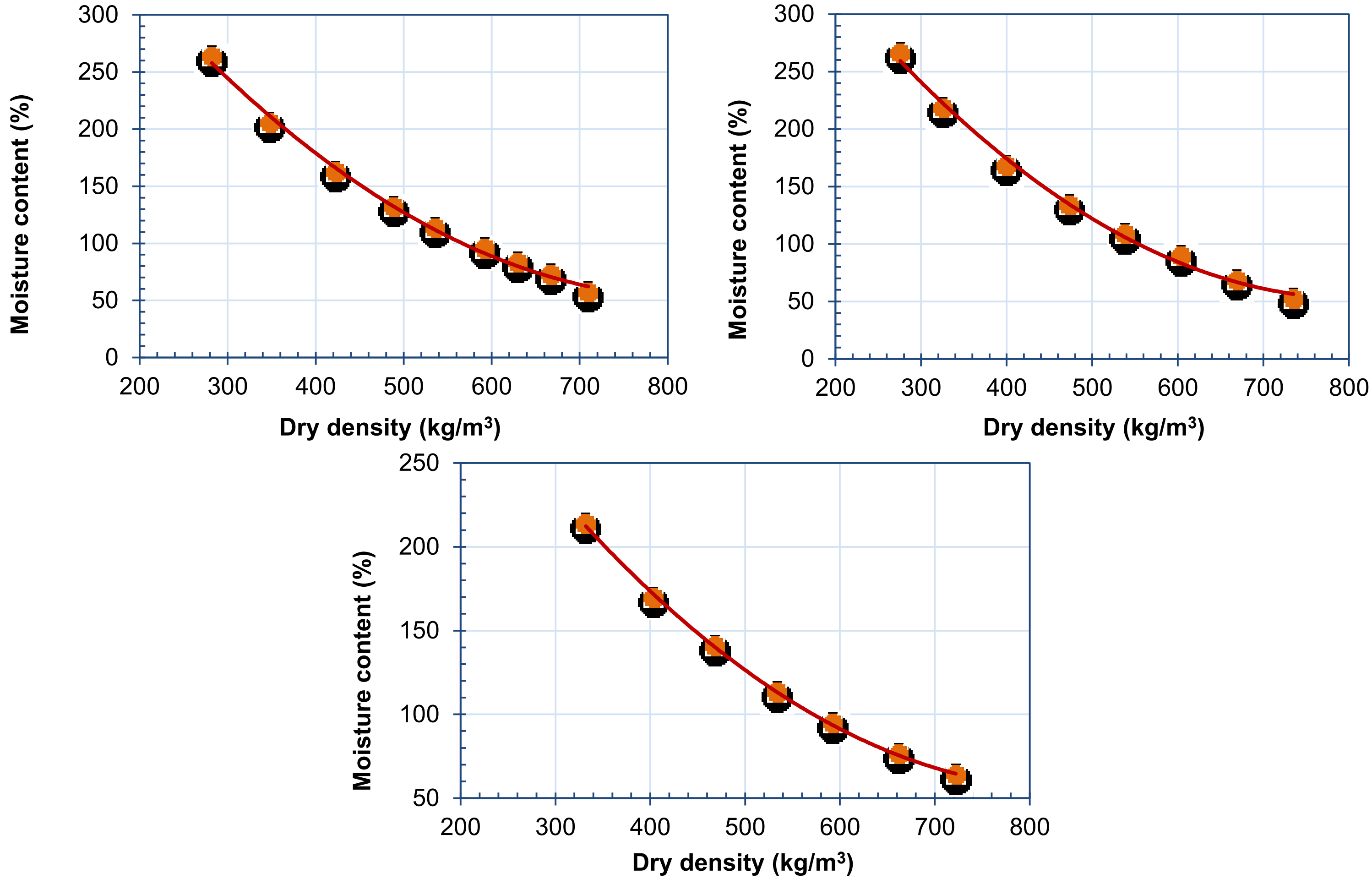


Figure 4-21 Variation of saturated M.C. With dry density for shredded waste-A from small, medium and large size devices

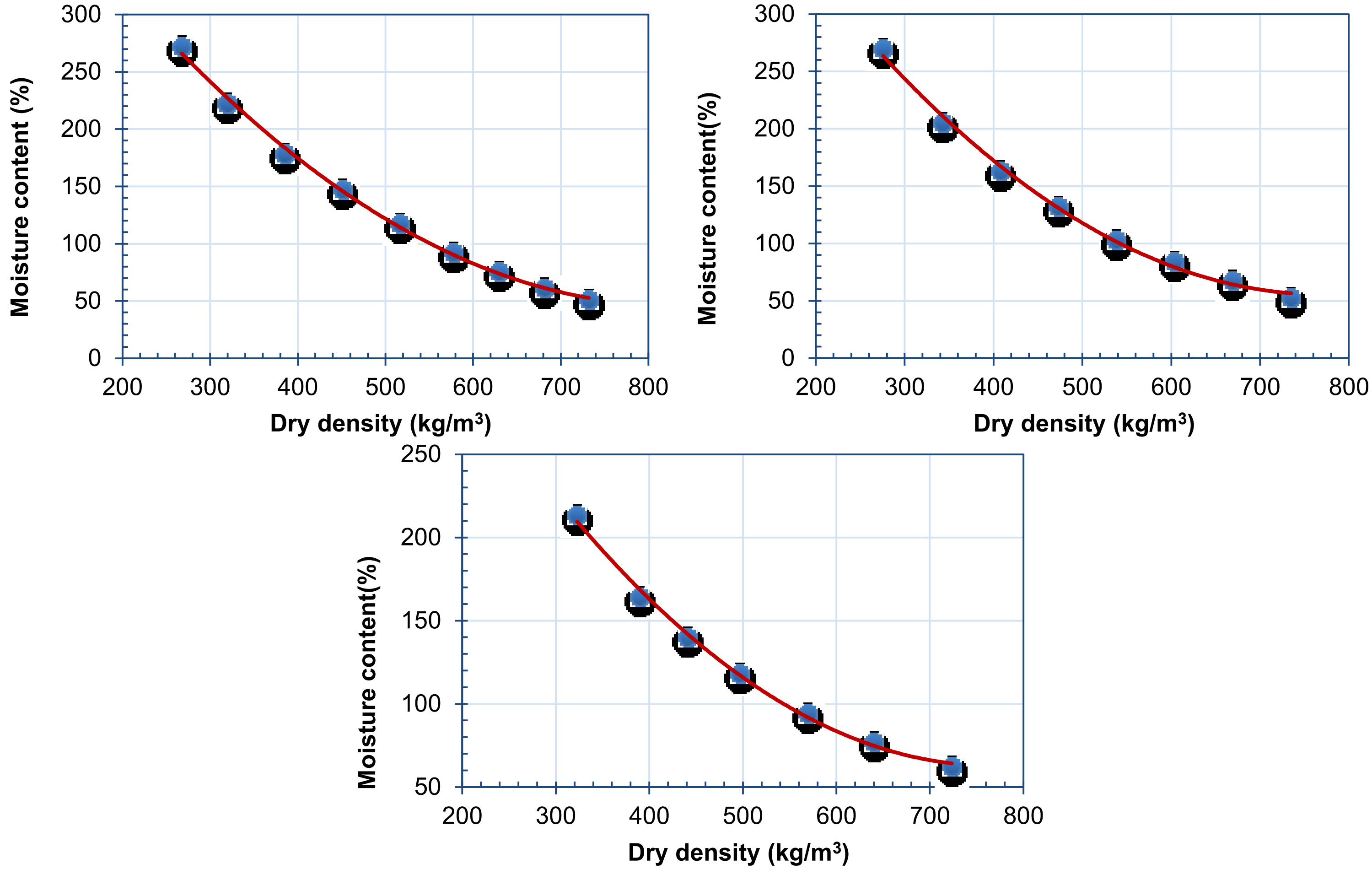


Figure 4-22 Variation of saturated M.C. With dry density for shredded waste-B from small, medium and large size devices

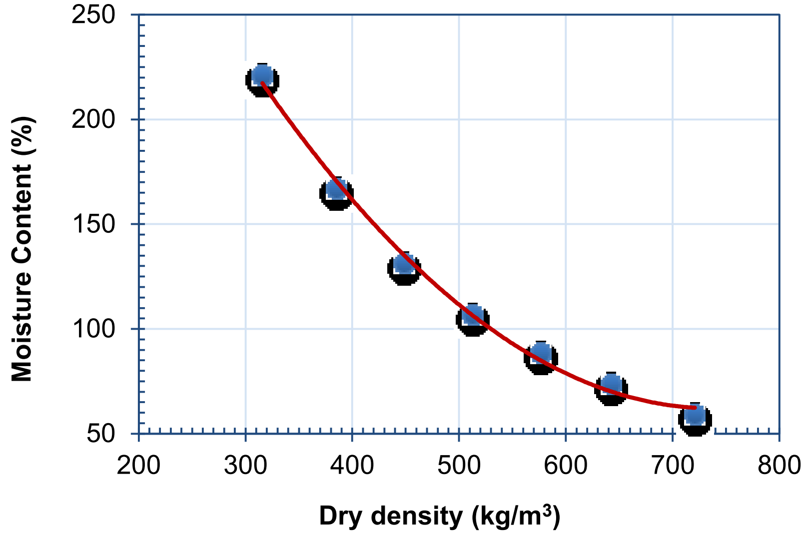


Figure 4-23 Variation of saturated moisture content with dry density for un-shredded waste-B obtained from large-scale device

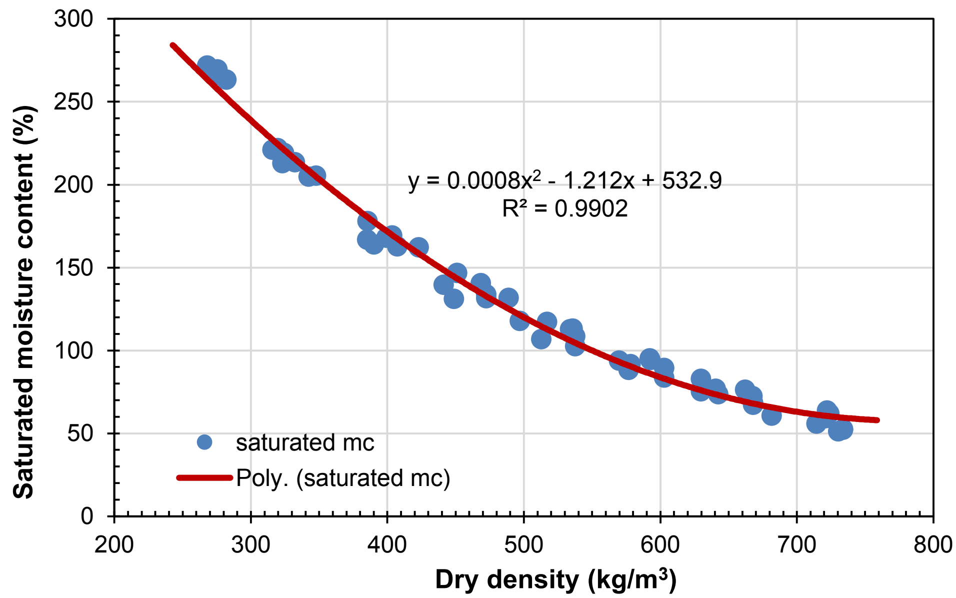


Figure 4-24 Relationship between saturated moisture content and dry density for fresh municipal solid waste

4.5.4.2 Retaining moisture capacity of municipal solid waste

Moisture contents were measured at total absorptive capacity for all waste samples at various densities. This is the maximum moisture content which solid waste can hold. The values obtained through laboratory tests were quite high as compared to field data. Generally there are many factors which can affect the moisture content in the field such as temperature, accumulation of huge amount of gas and unsaturation of waste. The waste can never get saturated in field because of not providing sufficient amount of water to saturate. It is also not possible to saturate the waste by applying water to waste in field. Application of water can cause instability and flooding which is undesirable for the landfill operation. Because of these various reasons, the moisture contents were always high for samples testing in laboratory scale as compared to field scale tests. The shredded waste-A has the maximum moisture retaining capacity ranging from 135.0 % to 51.3 % for the dry density between 281.9 kg/m^3 and 714.3 kg/m^3 while using a small size permeameter. Similarly, the waste-A has the value of moisture content ranging from 139.3 % to 48.7 % for the dry density between 275.3 kg/m^3 and 733.8 kg/m^3 while using a medium size permeameter. The large 25.4 cm diameter permeameter resulted in a range of moisture content from 119.2 % to 58.6 % for the dry density between 332.1kg/m^3 and 722.1 kg/m^3. The result of moisture contents for all waste-A samples are shown in Table 4-18, Table 4-19, Table 4-20 and in Figure 4-25.

Similarly, maximum moisture retaining capacity were measured at total absorptive capacity on all 25 waste-B sample. The shredded waste-B had retained moisture contents from 143.9% to 48.7% for the dry density between 267.8 kg/m^3 and 730.2 kg/m^3 while using a small size permeameter. Similarly, the waste-B had moisture contents from 135.8% to 47.5% for dry density between 275.3 kg/m^3 and 733.8 kg/m^3 using a medium size permeameter. The large 25.4 cm diameter permeameter resulted in a range of moisture contents of 125.6 % to 57.7 % for dry density between 322.9 kg/m^3 and 723.7 kg/m^3. The result of retained moisture contents for all waste-B samples are shown in Table 4-21, Table 4-22, Table 4-23 and in Figure 4-26. Similarly, maximum moisture retaining capacity for unshredded waste-B had ranging from 127.1% to 55.0 for dry density between 315.8 kg/m^3 and 720.9 kg/m^3 while using a large 25.4 cm diameter permeameter. The result of moisture contents for unshredded waste-B samples are shown in Table 4-24 and Figure 4-27

The results clearly demonstrated that the water retained capacity can be significantly influenced by increasing density. The retention capacity can also be affected by overburden stress and temperature. Overburden stress and temperature lower the water retention capacity of solid waste. Generally the water retention capacity will be low in the actual landfill because of the continuous application of overburden stress and temperature on waste mass. The absorptive capacity of MSW is an important parameter for bioreactor landfill. There might be several other factor affecting the absorptive capacity such as degradation and composition of waste. Water retention capacity is important for waste if the landfill operates as bioreactor. Higher water retention capacity increase the degradation process because of availability of water inside waste. While compacting the MSW, the volume of the voids in the waste will reduce so that water carrying capacity of the waste will also reduce. These results show there might be a correlation between the dry density and moisture content of MSW. The general trend is that the moisture content holding capacity decreases with increase in dry density of fresh MSW. Several others also reported the influence of density on the moisture content of MSW.

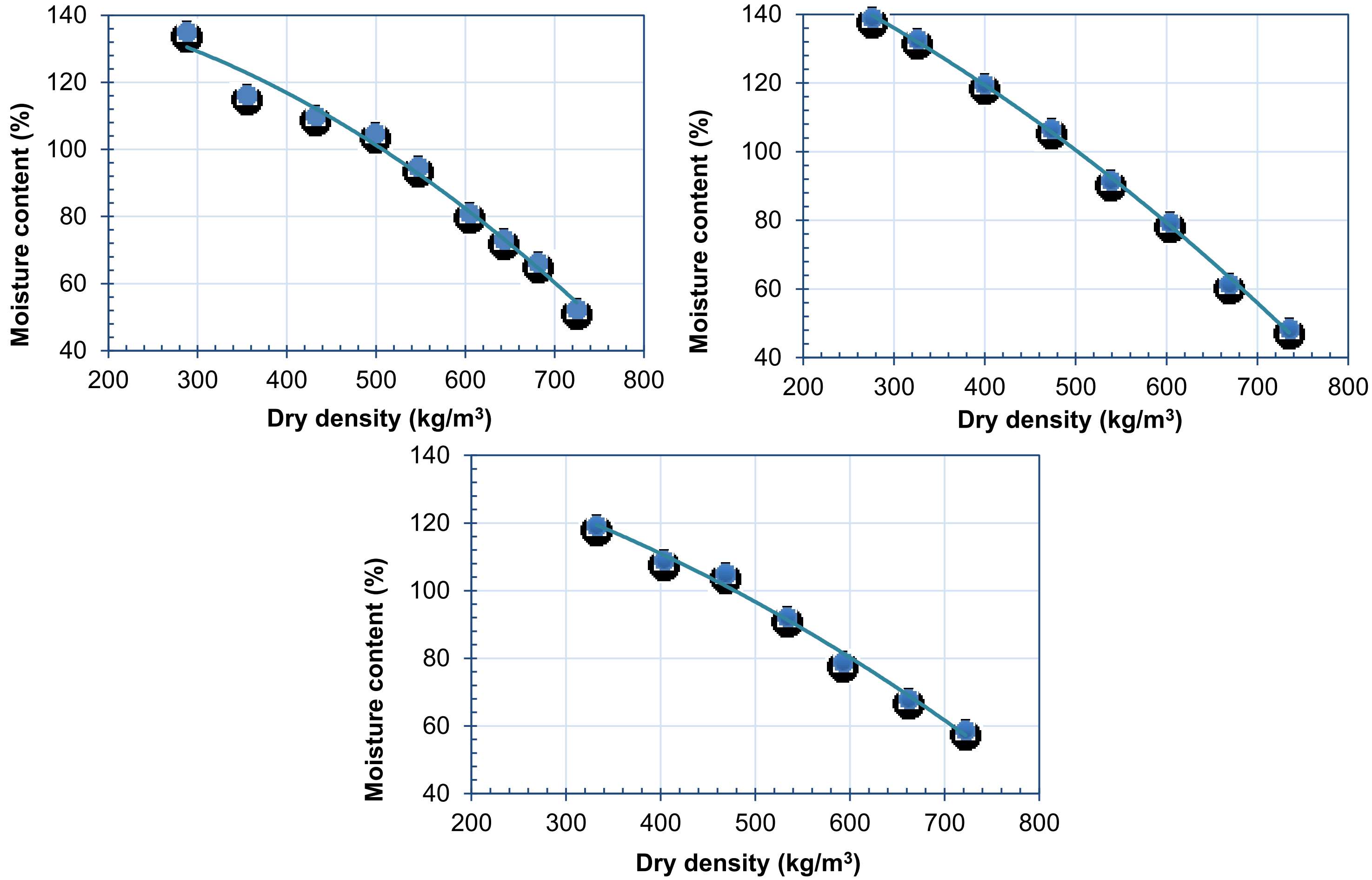


Figure 4-25 Variation of retained moisture content with dry density for shredded waste-A obtained from small, medium and large size devices

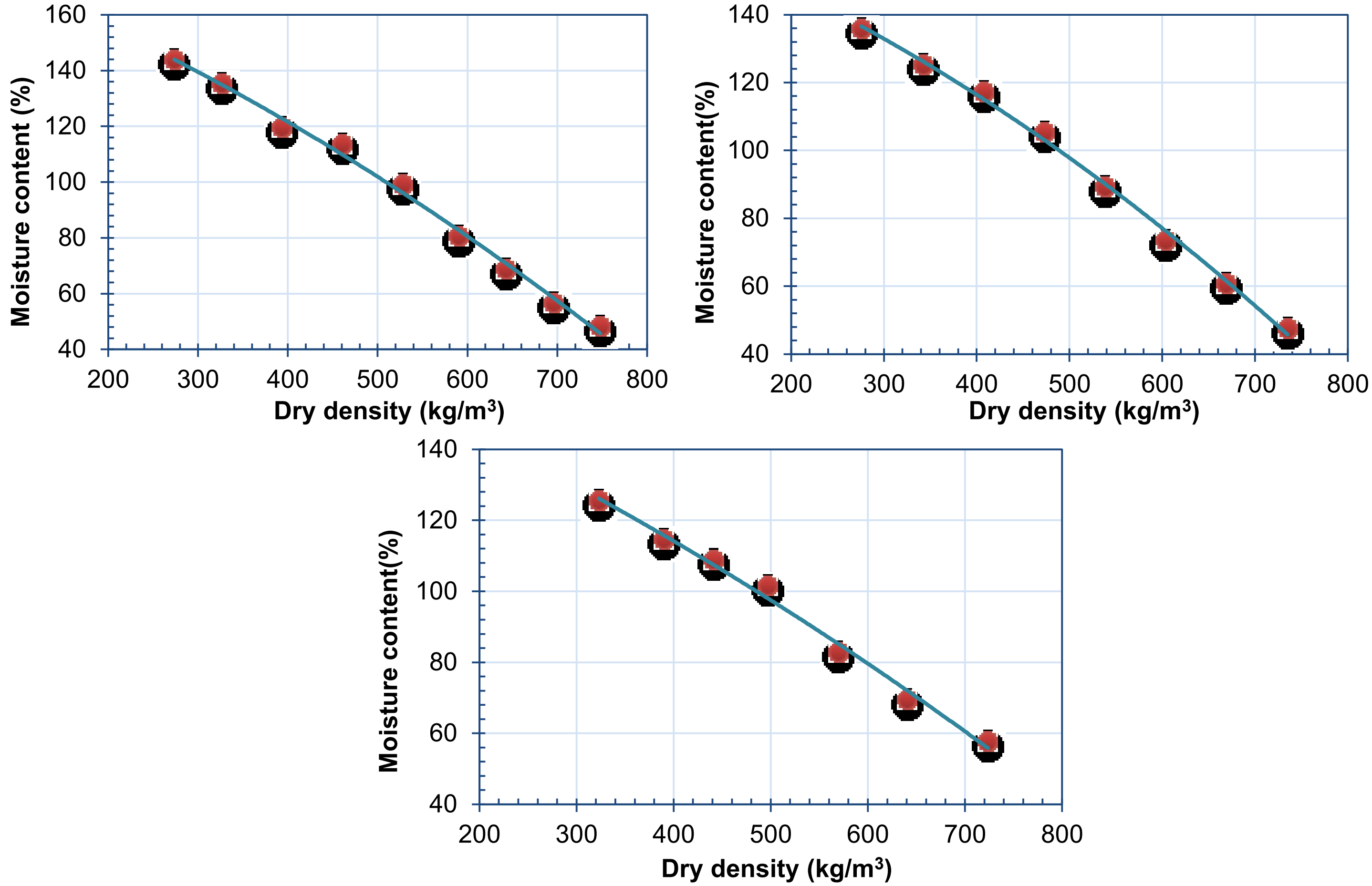


Figure 4-26 Variation of retained moisture content with dry density for shredded waste-B obtained from small, medium and large size devices

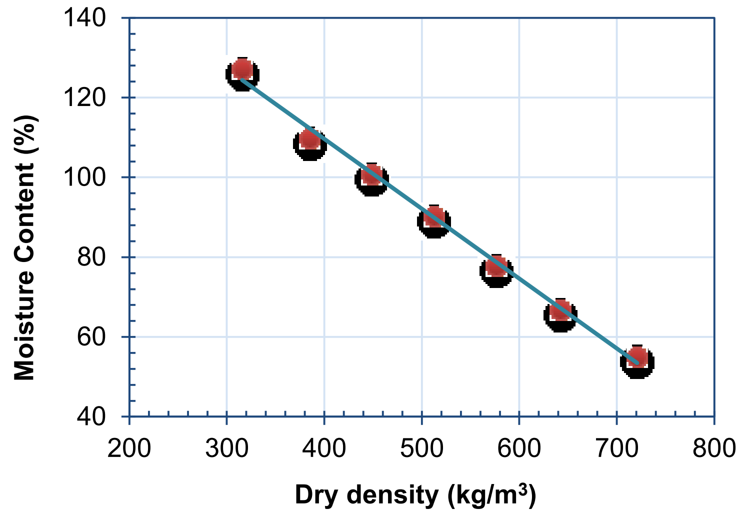


Figure 4-27 Variation of moisture content with dry density for un-shredded waste-B obtained from large-scale device

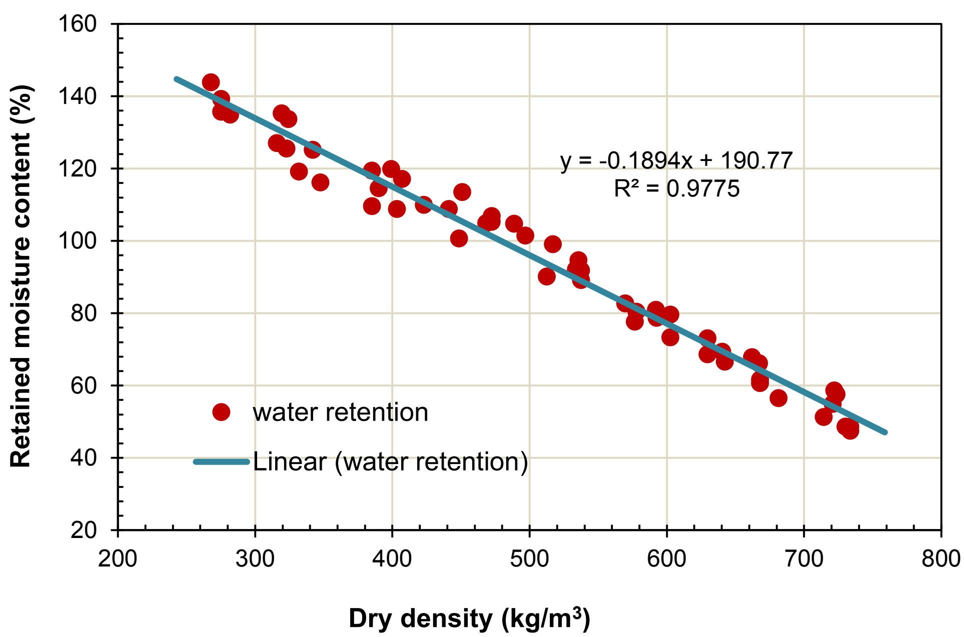


Figure 4-28 Relationship between moisture retention capacity and dry density for fresh municipal solid waste

When all the moisture retention capacity were plotted against dry densities for all of MSW samples, it followed a well-defined relationship between moisture retention capacity and dry density. The retained moisture contents were followed an equation with one degree order. The Figure 2-28 explains the relationship of water retention capacity with dry density. The correlation coefficient, R^2 (R-squared) of the equation was 98 % for the overall moisture retention capacities which were determined in the current research for various waste through different devices. The moisture retention capacity versus dry density equation can be obtained as,

$$y = -0.1894 \times x + 190.77 \ldots\ldots\ldots\ldots\ldots \text{(4.5)}$$

Where y is the water retention capacity (%) in percentage (%) and x is the dry density. The unit of the density is kg/m^3 while calculating saturated moisture contents. The equation (4.5) can be utilized to estimate moisture retaining capacity for the similar type of solid waste within narrow range of composition variation.

4.6 Comparison with previous work

Permeability of the MSW can be determined by laboratory tests and field tests. There were several studies available the hydraulic conductivity conducted on laboratory and field scale. A brief overview of comparison of current study to previous studies on permeability of MSW on undisturbed samples and reconstituted samples is presented in this section. Figure 4-30 gives a detail overview of previous results of permeability from previous various researchers and current values. The current values are considered here only from large permeameter of waste-A; waste-B and unshredded waste. All values are included from all stages of degradation.

The results clearly demonstrated that the hydraulic conductivity of MSW can be significantly influenced by the density. This is mainly attributed to the decrease in porosity and thus leading to the low void ratio. When confining pressure was increased in the landfill, it leads to the increase in density and thus lower the hydraulic conductivity There is a huge variation on the permeability of waste. The whole combined results literally does not show no a definite correlation between the dry unit weight and hydraulic conductivity. It only concludes the permeability is greatly influenced by the increasing in density.

The values of hydraulic conductivities of MSW reported in the literature vary between approximately $1X10^{-1}$ cm/s and $1X10^{-7}$ cm/s. However, simply stating a range of hydraulic conductivities of waste will not be particularly helpful because the range is potentially very wide. The variation might be due to the effect of factors such as waste composition and density which influence the hydraulic conductivity in an understandable and perhaps even predictable way. Besides these, the permeability was greatly influenced by the air content inside the permeameter and waste. Generally here will be continuous generation of gas inside waste due to water. As the gas generation was greatly increased by the water content, the permeability will lower due to accumulation of gas inside waste.

The permeability of MSW with density of the current study is presented in Figure 4-29. The test results showed that the permeability decreased with increasing density for all wastes. Blieker et al. (1993) reported difficulties in drilling on landfill with depth, which implies the increase in density with depth. Oweis and Khera (1990); Blieker et al. (1993); Brandle (1994); Chen and Chyonoweth (1995); Powrie et al. (2005), Hossain et al., (2009); Staub et al., (2009); Reddy et al., (2009) indicated decreasing in permeability with increasing in density. The experimental results from current study conform to the existing research.

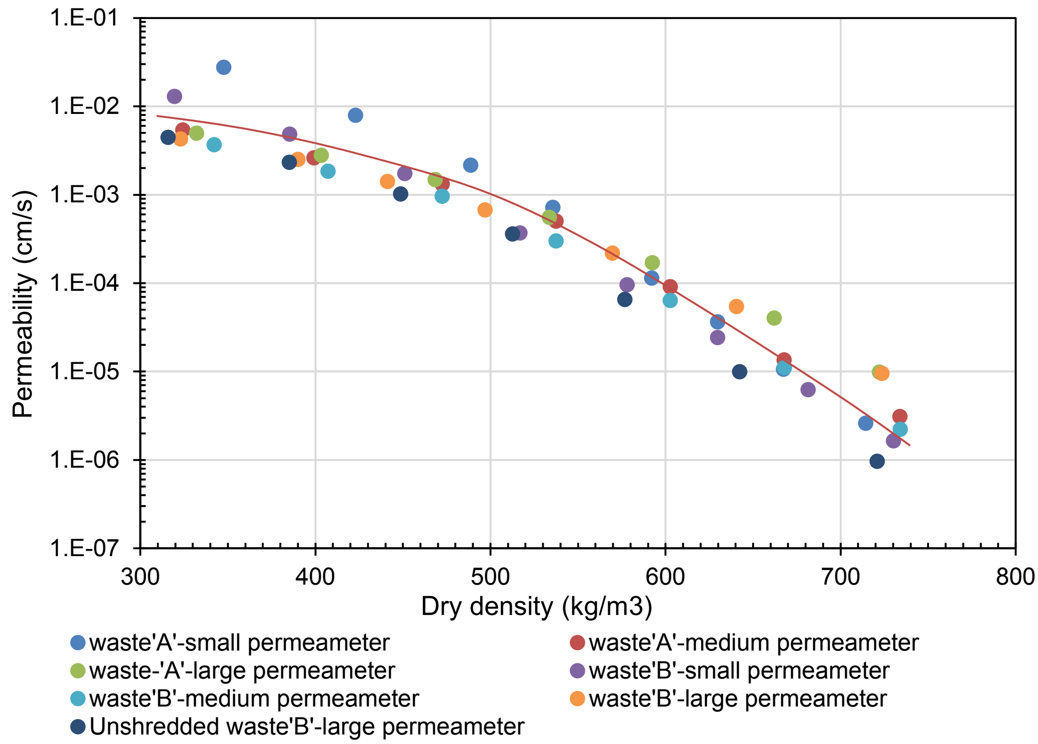


Figure 4-29 Summary of relationships between log10 vertical hydraulic conductivity and waste dry density

The Figure 4-30 gives the comparison of results obtained in the current research with permeability obtained by other various researchers. The permeability results when plotted against their corresponding densities gave an agreeable relation with the test results of these researchers. Generally, most of hydraulic conductivities have determined in vertical flow in the laboratory whereas in a landfill the anisotropy resulting from the deposition and layering of the waste would be expected to result in preferential horizontal flow (Beaven et al., 2011). There are numerous factors which caused complexity in permeability in landfills. The complications begin from the effects of channeling and preferential flow, gas generation, random heterogeneity and waste processing, decomposition, cover soil layer, composition and compaction effort. As illustrated in

Figure 4-29, the relationships between hydraulic conductivity and the dry density appear to be well defined and slightly varying for individual waste types. In this current study, the composition of the waste are same for all samples for individual wastes types. As the previous data are scattered in wide range, it is determined to make consistency in the composition and varying only the density. One of the primary cause of differing in values is the composition. Many authors have presented data relating hydraulic conductivity to waste density, but it is hardly to conform whether the composition were exactly similar for their samples. Another reason might be the particle sized used by authors in their corresponding research. It is also one of the difficult task getting the composition of the waste used by previous researchers. Actually, the effects of waste composition and other variables make estimation of permeability such a relationship to different wastes and other situations rather difficult and complex.

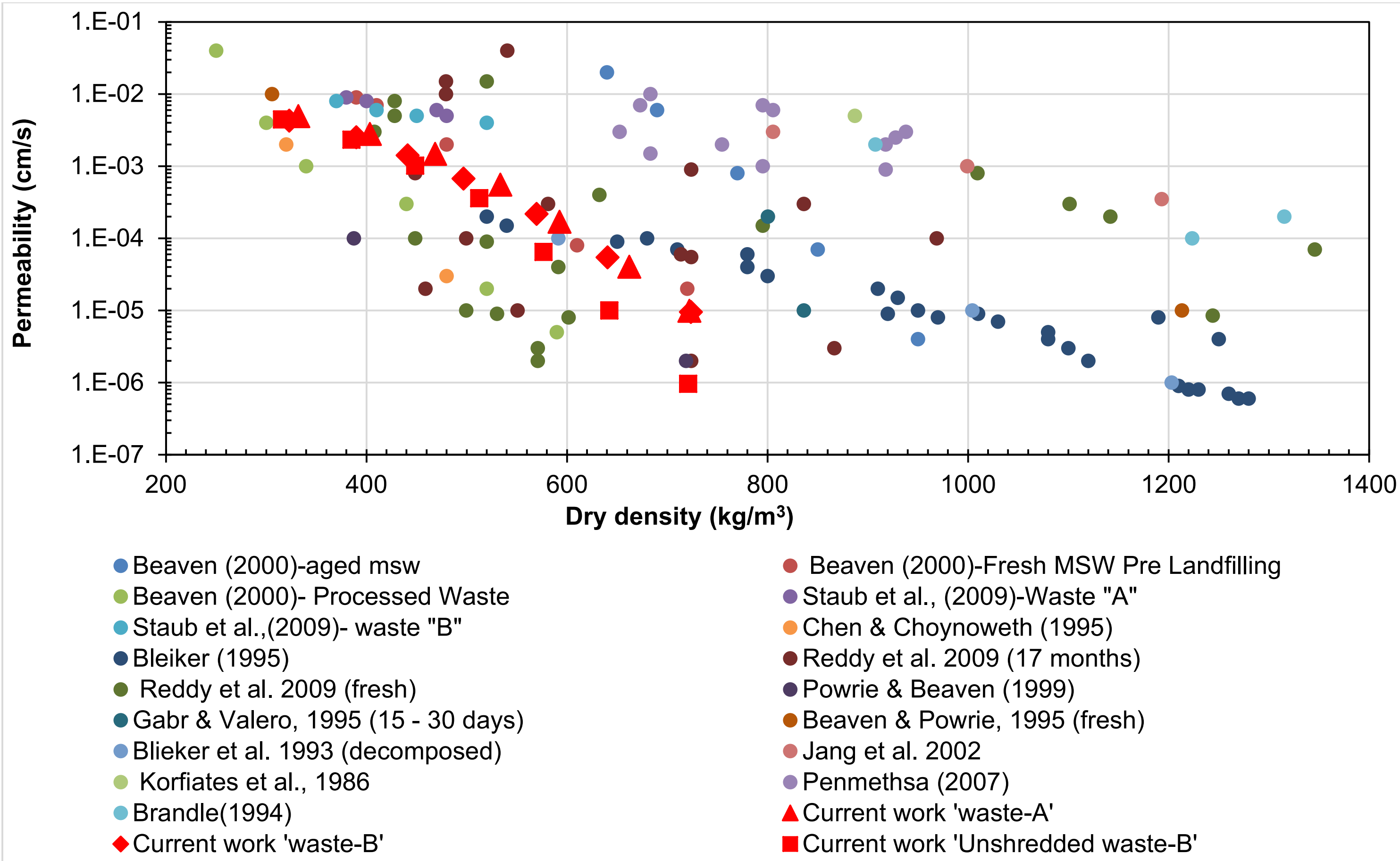


Figure 4-30 Summary of relationships between log10 vertical hydraulic conductivity and waste dry density

4.7 Factor affecting permeability of municipal solid waste

Permeability in municipal solid waste is mainly dependent on the density, pore size and its geometry but it might be affected by various factors which are discussed in this chapter. There was not a standard procedure to determine the permeability of solid waste. There was not any specific criteria for selecting the waste particle, size and shape. As the MSW is very heterogeneous material, its properties are also not uniform through the waste mass. It is mainly due to composition and particle size variation. The individual components of solid waste have different properties. Some components have high permeability and some have very low permeability so that the variation on amount of those components makes difficult to understand the exact flow behavior of solid waste. If the composition and density are also same, the selection of equipment, size of sample, size of waste particle might affect the hydraulic conductivity of MSW. For these reason, knowledge of the hydraulic conductivity of municipal solid wastes and an understanding of the factors that control it are essential. Several research had already focused on the effect of density and porosity on the hydraulic conductivity. The effect of density and porosity had already mentioned in previous sections. It is tried to focus to those parameter which have very limited information in the literature.

4.7.1 Initial moisture content at compaction

Waste having same kind of composition was considered to perform the effect of initial moisture content on the permeability. Moisture contents were varied from 10% to 70% in the 20% interval and two densities 402.9 kg/m^3 and 515.2 kg/m^3 were selected to observe the variation on permeability of waste. Hydraulic conductivity decreased with increasing moisture content as shown in Figure 4-31. The ratios of maximum to minimum hydraulic conductivities were 2.56 and 3.22 in densities 402.9 kg/m^3 and 515.2 kg/m^3, respectively. Similar kind of trends were observed in both density. This might be due to decreasing porosity and affecting the path of flow due to moisture content. When waste is compacted in wet condition, there might be air entrapped inside the waste. The entrapped air might break the uniform path of water and also lower the permeability.

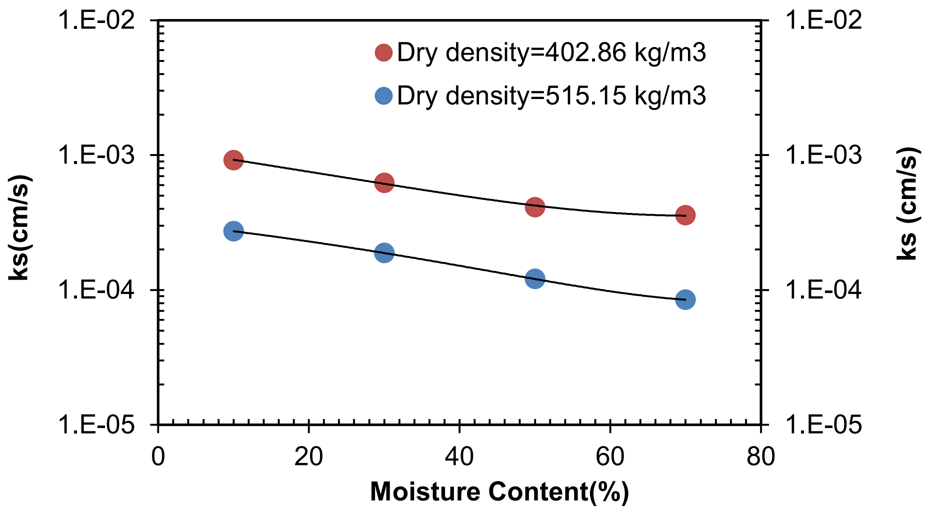


Figure 4-31 Effect of moisture content on the hydraulic conductivity

4.7.2 Hydraulic head (Δh)

Waste having same kind of composition was considered to perform the effect of hydraulic head on the permeability of waste. Hydraulic heads were varied from 84.8 cm to 203.2 cm and three densities 320.4, 400.5 kg/m^3 and 480.6 kg/m^3 were selected to observe the variation on permeability of waste. Hydraulic conductivity was slightly increasing with increasing hydraulic head as shown in Figure 4-32. The ratios of maximum to minimum hydraulic conductivities were 1.39, 1.45 and 1.91 when the hydraulic head decreased from 203.2 cm to 84.8 cm in three densities 320.4 kg/m^3, 400.5 kg/m^3 and 480.6 kg/m^3, respectively. Similar kind of trend were observed in all three dry densities. This might be due to creating of water channel due to higher pressure. Higher head resulted higher pressure inside the permeameter which might create flow channel. The hydraulic conductivity is expected to be higher due to channeling effect.

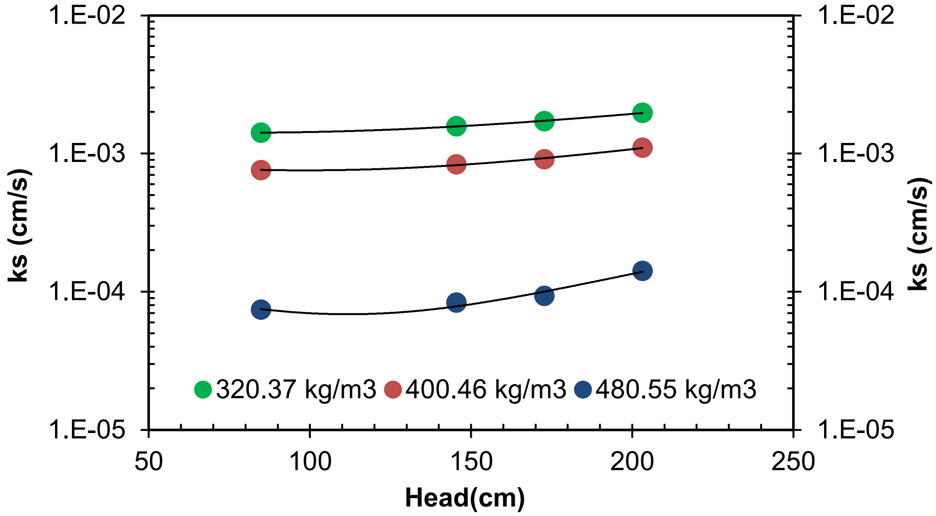


Figure 4-32 Effect of hydraulic head on the hydraulic conductivity

Chen and Chynoweth (1995) had performed the effect of hydraulic gradient on the permeability of the variable waste. They observed the hydraulic conductivity of MSW was mostly influenced by packing density. It was not much affected by hydraulic gradient. It can be affected by the composition of MSW made up of paper, plastic and yard waste.

4.7.3 Particle size of waste

Waste having same kind of composition was considered to perform the effect of particle size on the permeability of waste. Permeability tests were performed in the same 15.24 cm diameter permeameter in identical condition. Wastes components were shredded into average size of 2.54 cm, 6.35 cm and 10.15 cm. All components were mixed together and compacted into permeameter. In order to determine the true effect of particle size on the permeability of MSW, three densities 320 kg/m^3, 400 kg/m^3 and 480 kg/m^3 were considered. Hydraulic conductivity was plotted against to the ratio of particle size to equipment diameter. Hydraulic conductivity was slightly decreasing with increasing the ratio which is shown in Figure 4-33. The ratios of maximum to minimum hydraulic conductivities are 3.32, 2.30 and 2.00 when the particle size decreased from

10.15 cm to 2.54 cm in three densities 320 kg/m^3, 400 kg/m^3 and 480 kg/m^3, respectively. Similar kind of trend were observed in all three dry densities. This might be due to creating of more water channels due to smaller size of waste particles. When the particle is shredded into number of small particles, the surface area is increased and thus void ratio. This creates the path on the waste and increased the flow.

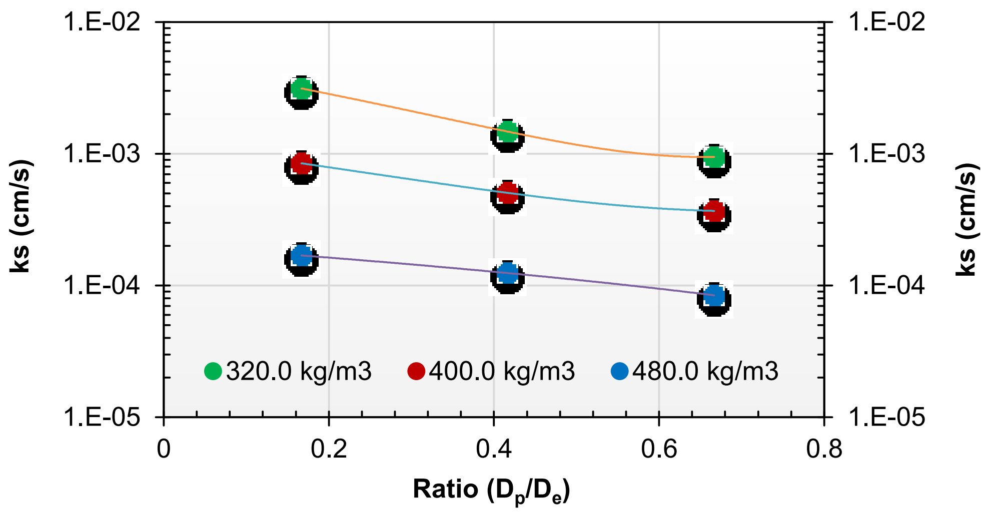


Figure 4-33 Effect of particle size on the hydraulic conductivity

4.7.4 Effect of shredding

Waste having same kind of composition was considered to perform the effect of shredding on the permeability of waste. Permeability tests were performed in the same large permeameter in identical condition. Similar waste was considered to conduct the tests. In one series of tests, wastes were taken without shredding in natural state while waste components were shredded into average size 10.15 cm in another varies of tests. All components were mixed together and compacted into permeameter. In order to determine the true effect of shredding on the permeability of MSW, 14 tests were performed at various densities. The values of hydraulic conductivity were bigger in shredded waste in all densities which are shown in Figure 4-34. Although the difference in hydraulic conductivity were smaller in below 500 kg/m^3 but it increased around 10 times in shredded waste as compared to unshredded waste in high dry density 720 kg/m^3. This might be due to creating of more water channel due to smaller size of waste particles. When the particle is shredded into number of small particles, the surface area is increased and thus void ratio also increased. This creates the water path on the waste materials and increased the flow.

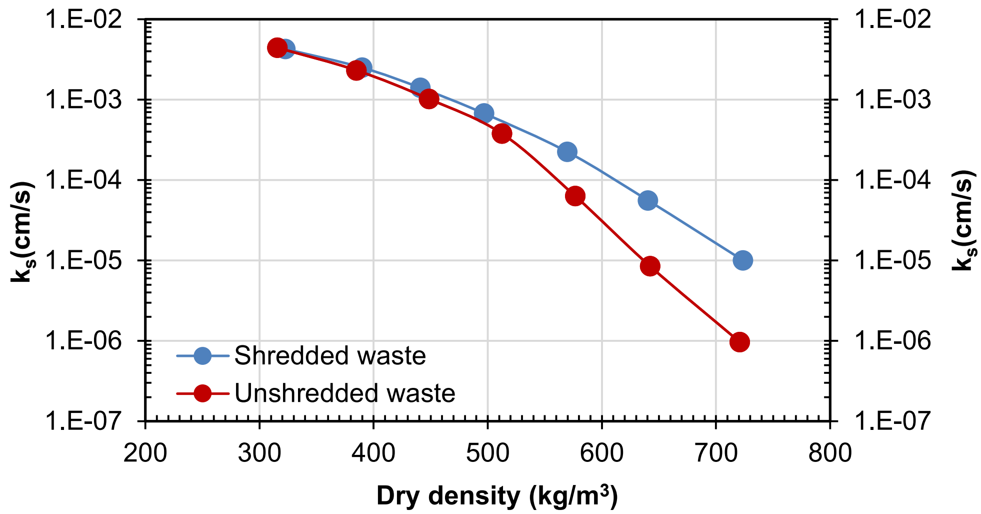


Figure 4-34 Effect of shredding on the hydraulic conductivity

4.7.5 L/r ratio of waste sample

Waste having same kind of composition was considered to perform the effect of ratio of length of waste sample to the radius of sample on the permeability of waste. Permeability tests were performed in the same large permeameter in identical condition. Waste having same kind of composition was considered to perform the effect of length/radius on the permeability of waste. All 5 tests were conducted on the same density and composition. Hydraulic conductivity was slightly decreasing with increasing length to radius ratio. The minimum to maximum ratio of hydraulic conductivities is 0.61 when the L/r ratio increased from 0.2 to 1.07 which is shown in Figure 4-35.

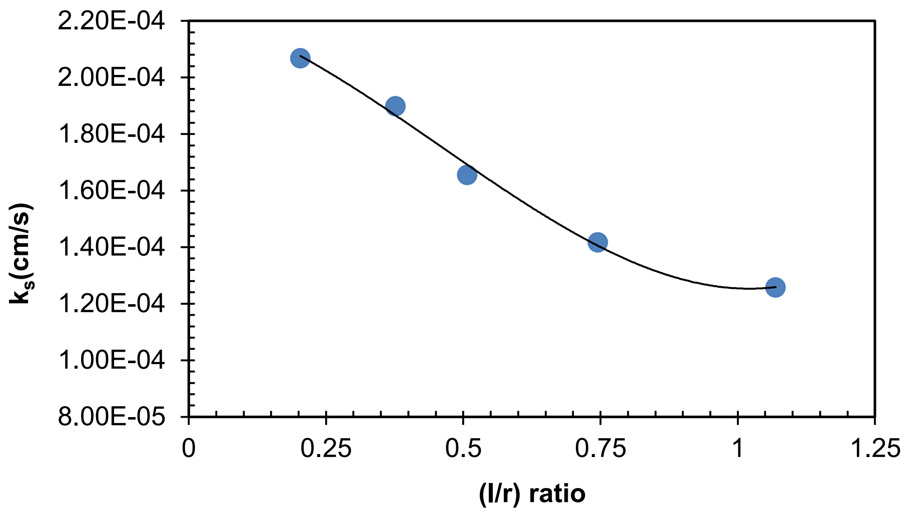


Figure 4-35 Effect of length to radius of sample on hydraulic conductivity

4.8 Degradation of municipal solid waste

There are several factors which can influence the degradation process within the landfill. These parameters include stress/density, moisture content, nutrient content, pH level, bacterial content, oxygen concentration, and temperature. There has been limited research available in the degradation process. Since the density is one of the most important parameters in landfill, it has been focused the effect of density on the degradation of the MSW. These factors alone may not be critical, however, they may influence other parameters that control municipal solid waste degradation rates and activities (Pohland 1995).

4.8.1 Gas generation

MSW samples with the same physical characteristics were filled in the three bioreactor cells at dry density of 457.8 kg/m^3, 572.1 kg/m^3, 686.3 kg/m^3 which were designated as R1, R2 and R3 respectively. The initial moisture content was maintained at 40% on dry basis while on compaction. The cells were equipped also with leachate collection, leachate recirculation, gas collection along with capability of performing permeability tests. This subsection covers rate of methane production, cumulative methane and concentrations produced from the three different compacted MSW samples. Gas generation was monitored from reactors almost one year in all three compacted MSW samples.

4.8.1.1 Gas generation from reactor R1 (458.5 kg/m^3)

The reactor R1 has the lowest dry density of the MSW sample which was 457.8 kg/m^3. The most important parameters to observe the degradation is methane gas generation from the reactor. The generation of methane gas was almost negligible up to 30 days of reactor operation because composition of methane percentage was very low during those time. After 25 days the composition of methane percentage on gas started to increase and reached to around 50% of the total gas production as shown in Figure 4-36. The peak of the methane gas percentage was around 58% on after 150 days. After 90 days of bioreactor cell operation, the methane percentage reached almost constant i.e. 55% of the total gas generation. The maximum methane generation rate was 320 mL/kg/day on the 150 days. After 150 days there was decrease on the methane generation rate. The average rate of methane generation was around 220 mL/kg/day after 150 days and remain constant around up to 280 days of reactor operation. Methane generation rate started to decrease from 280 days of bioreactor cell operation. The total gas generation and methane gas generation rate are clearly explained in Figure 4-37. The relationship between methane gas generation rate and the variation of pH is explained in the Figure 4-38. Basically when the pH increased to 6.6 the methane gas generation started to increase. The methane generation rate reached to maximum when the pH almost reached maximum. The total methane generation was 52.1 L/kg of the MSW in this lowest compacted samples. Reactor R1 produced 109.6 L/kg of the total gas within 363 days. The ratio of the methane gas to the total gas generation was 47.56%.

The relationship between total gas and methane gas generation rate are explained on the Figure 4-39.

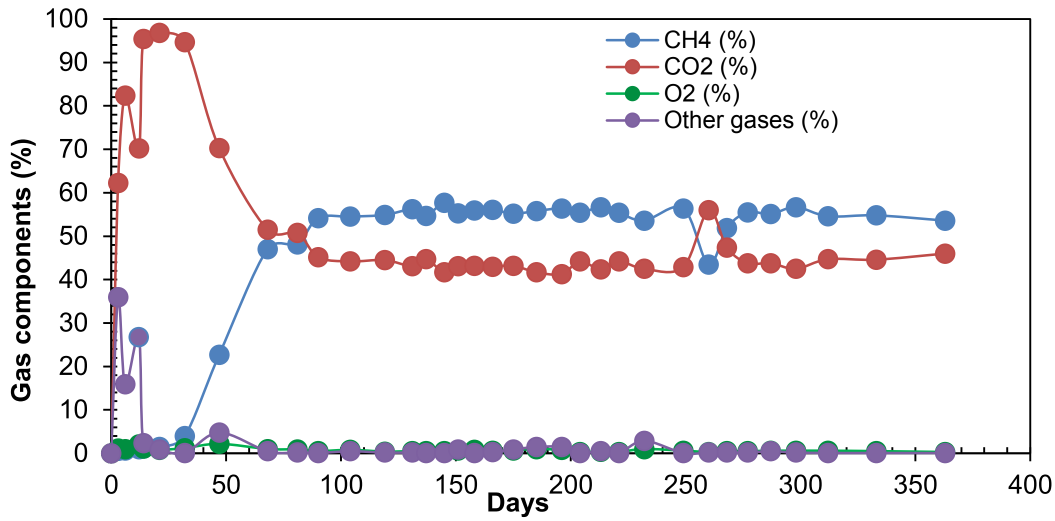


Figure 4-36 Composition of the various gases at different time in Reactor R1 compacted at density 457.8 kg/m^3

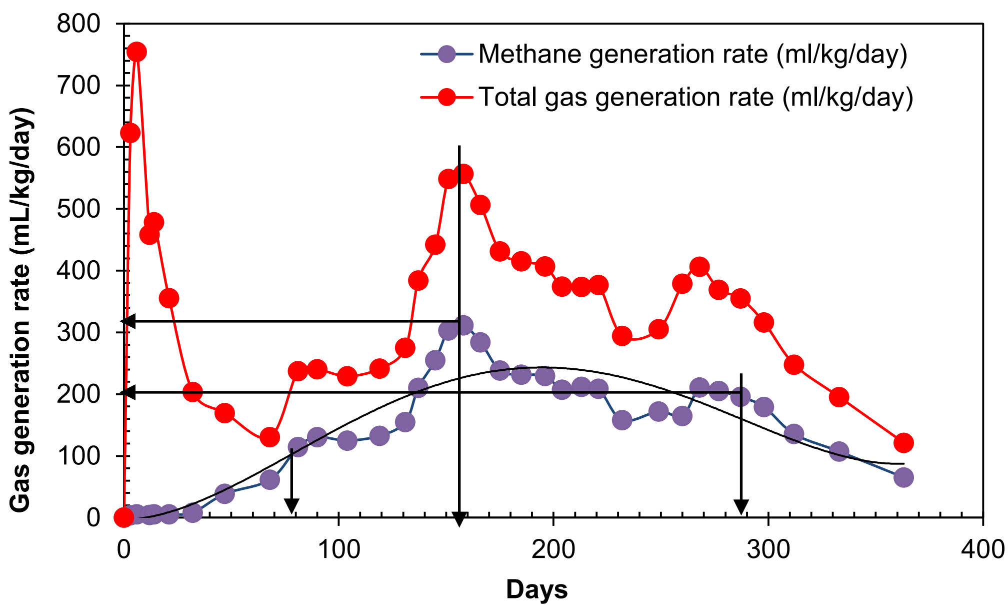


Figure 4-37 Total gas and methane gas generation rates in reactor R1 compacted at density 457.8 kg/m^3

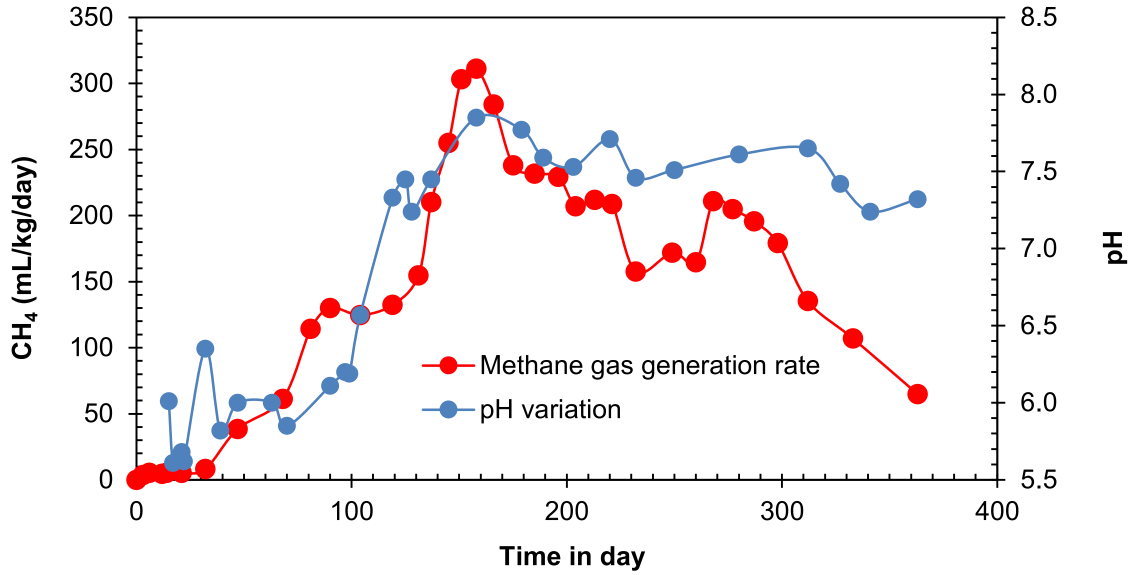


Figure 4-38 Relationship of methane generation and pH variation on reactor R1 compacted at density457.8 kg/m^3

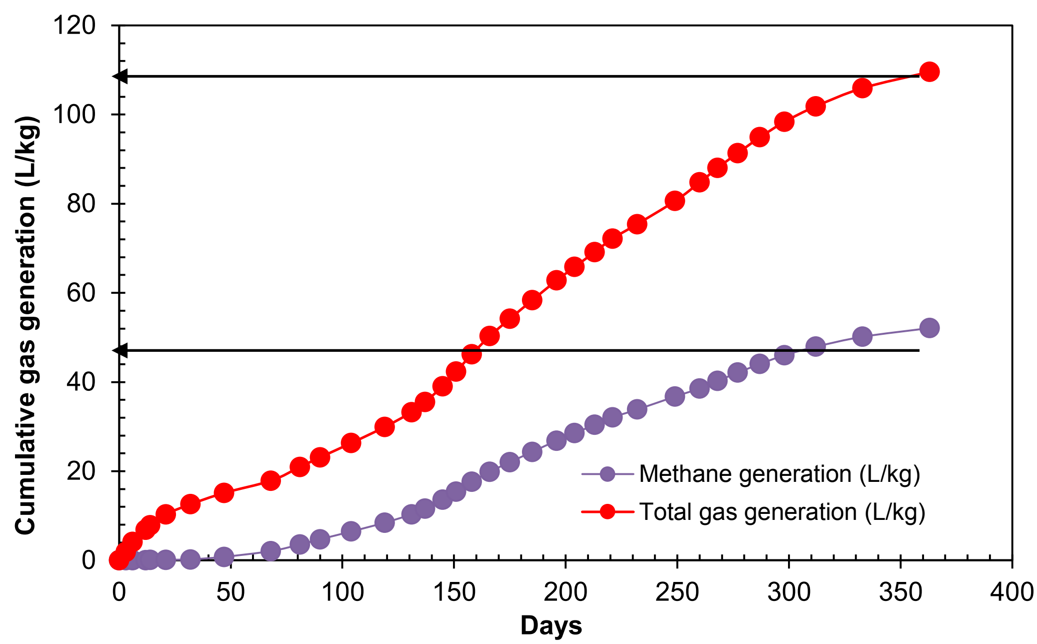


Figure 4-39 Cumulative total gas and methane gas generation in reactor R1 compacted at density 457.8 kg/m^3

4.8.1.2 Gas generation from reactor R2 (572.1 kg/m^3)

The reactor R2 has the dry density of the MSW sample which was 572.1 kg/m^3. The methane generation rate was almost negligible up to 50 days of reactor operation and the percentage of methane composition on the total gas was also low during these time period. After 70 days of operation the amount of methane percentage on gas started to increase and reached to around 44% of the total gas production as shown in the Figure 4-40. The peak of the methane gas percentage was around 56% on after 150 days. After 95 days of bioreactor cell operation, the methane percentage reached almost constant i.e. 55% of the total gas generation. The maximum methane generation rate was 245 mL/kg/day on the 150 days. After 150 days there was decrease on the methane generation rate. The average rate of methane generation was around 180 ml/kg/day after 150 days and remain constant around up to 280 days of reactor operation. Methane generation rate started to decrease from 280 days of bioreactor cell operation. The total gas generation and methane gas generation rate are explained in Figure 4-41. The relationship between methane gas generation rate and the variation of pH is explained in the Figure 4-42. Basically when the pH increased to 6.6 the methane gas generation started to increase. The methane generation rate reached to maximum when the pH almost reached maximum. The total cumulative methane generation was 39.3 L/kg of the MSW from R2 reactor. Reactor R2 produced 84.8 L/kg of the total gas within 363 days. The ratio of the methane gas to the total gas generation was 46.34%. The cumulative generation of total gas and methane gas generation rate are explained on the Figure 4-43.

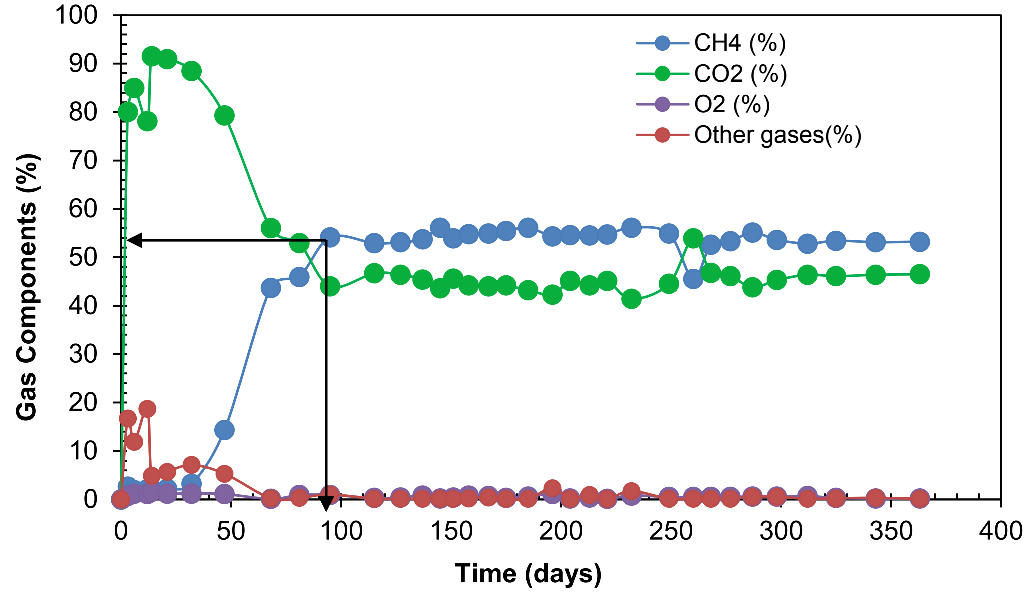


Figure 4-40 Composition of the various gases at different time in Reactor R2 compacted at density 572.1 kg/m^3

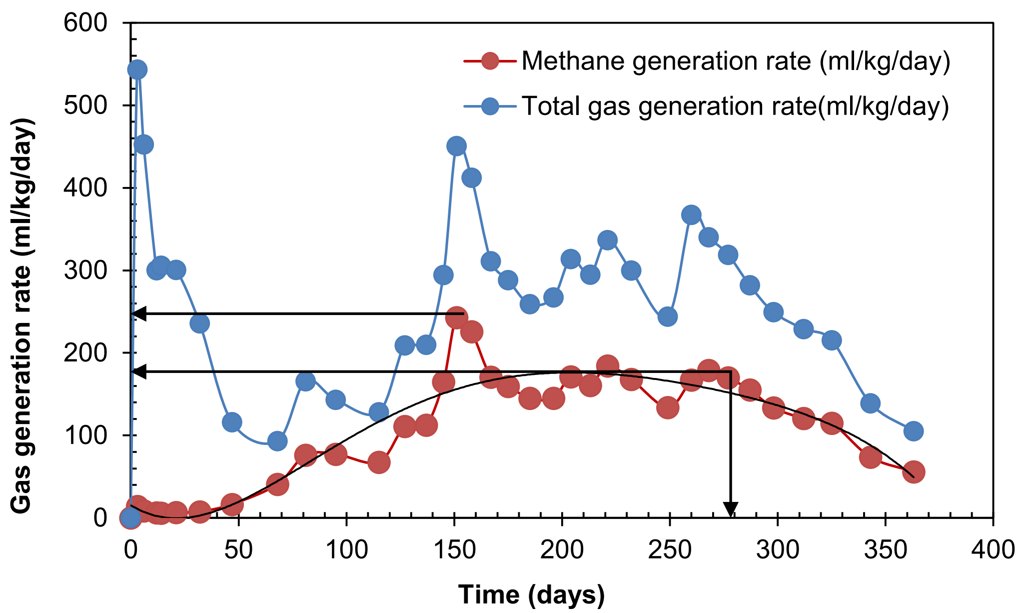


Figure 4-41 Total gas and methane gas generation rates in reactor R2 compacted at 572.1 kg/m^3

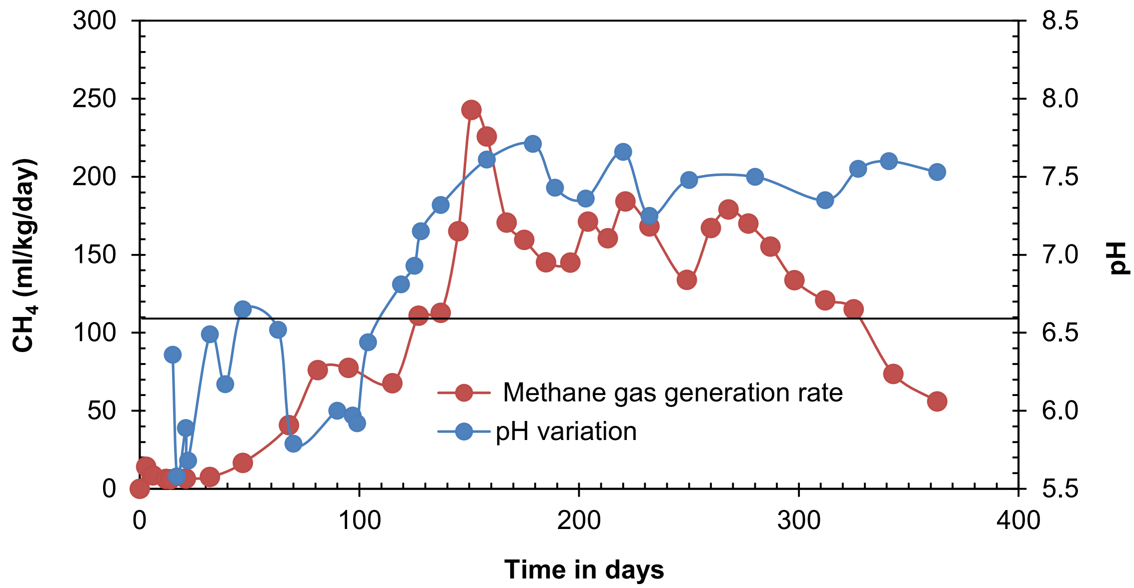


Figure 4-42 Relationship of methane generation and pH variation on reactor R2 compacted at 572.1 kg/m^3

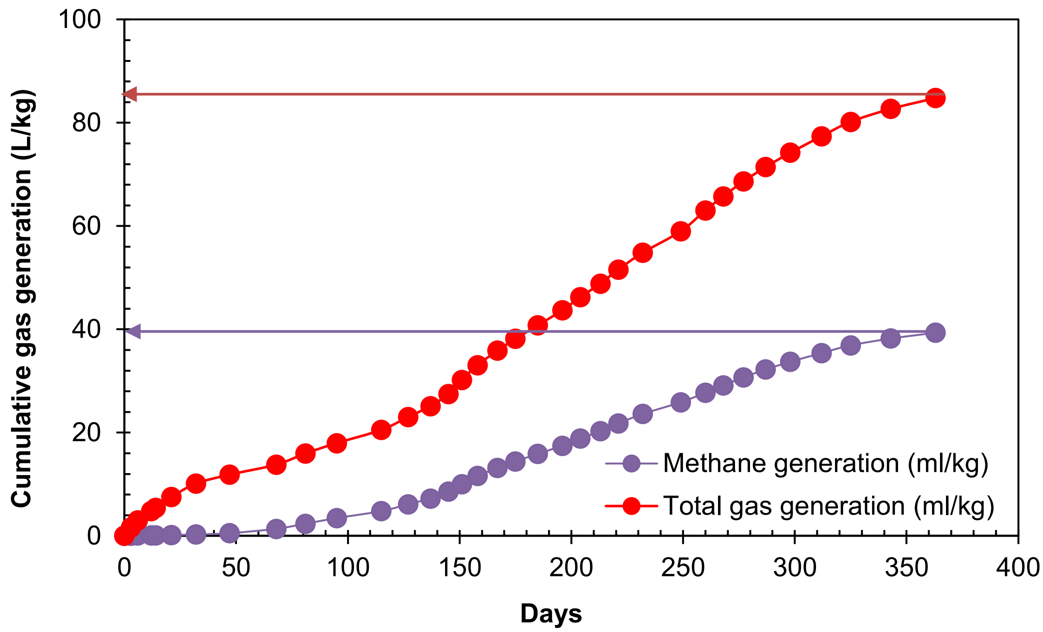


Figure 4-43 Cumulative total gas and methane generation in reactor R2 compacted at density 572.1 kg/m^3

4.8.1.3 Gas generation from reactor R3 (686.3 kg/m^3)

The reactor R3 had the highest dry density of the MSW sample which was 686.3 kg/m^3. The most important parameters to observe the degradation is methane gas generation from the reactor. The gas generation from reactor R3 also had been monitored continuously for 363 days. The generation of methane gas was almost negligible on whole duration of reactor operation even though the composition of methane percentage was high. The composition of methane percentage on gas started to increase after 70 days of reactor operation and reached to around 41% at the 70th day as shown in Figure 4-44. The peak of the methane gas percentage was around 60%. After 140 days of bioreactor cell operation, the methane percentage reached almost peak and remain constant for a long period of time constant. The maximum methane generation rate was 90 mL/kg/day on the 140th and 270th days of reactor cell operation which was very low as compared to other reactors values. The average rate of methane generation was around 75 mL/kg/day. Methane generation rate started to decrease from 270 days of bioreactor cell operation. The total gas generation and methane gas generation rate are clearly explained in Figure 4-45. The relationship between methane gas generation rate and the variation of pH is explained in the Figure 4-46. Basically when the pH increased to 6.6 the methane gas generation was slightly more as compared to other pH ranges. There was no distinct relationship between variations of pH with the methane generation rate. The generation of methane gas was negligible as compared to other reactors value and also with previous researchers' values. The total methane generation was 12.5 L/kg of the MSW in this lowest compacted samples. Reactor R1 produced 31 L/kg of the total gas within 363 days. The ratio of the methane gas to the total gas generation was 40.32%. The relationship between total gas and methane gas generation rate are explained on the Figure 4-47.

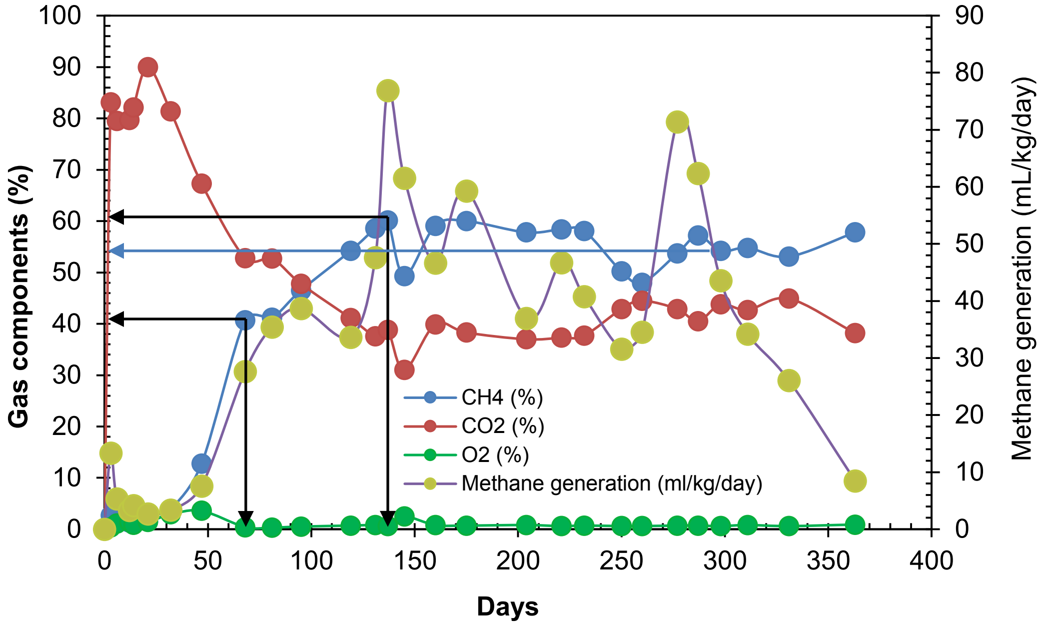


Figure 4-44 Composition of the various gases at different time in reactor R3 compacted at the density 686.3 kg/m^3

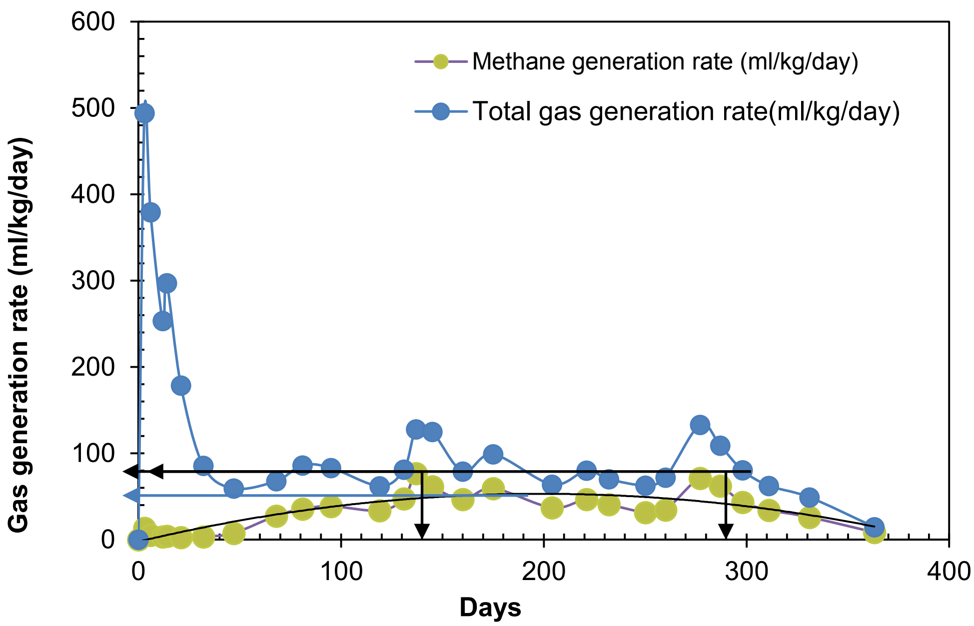


Figure 4-45 Total gas and methane gas generation rates in reactor R3 compacted at density 686.3 kg/m^3

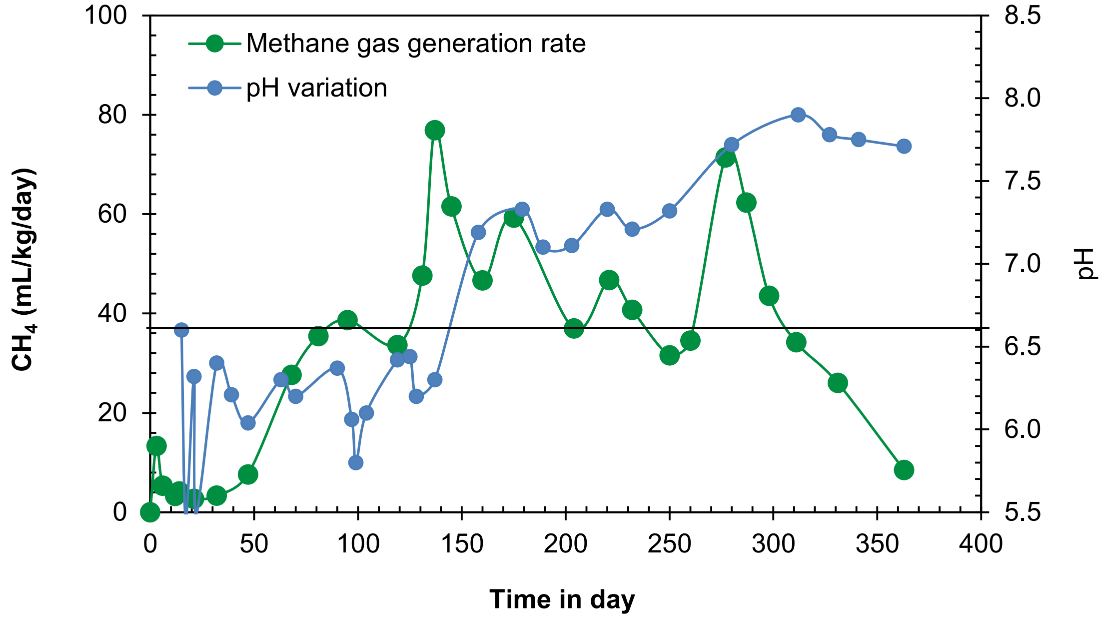


Figure 4-46 Relationship of methane gas generation and pH variation on reactor R3 compacted at density 686.3 kg/m^3

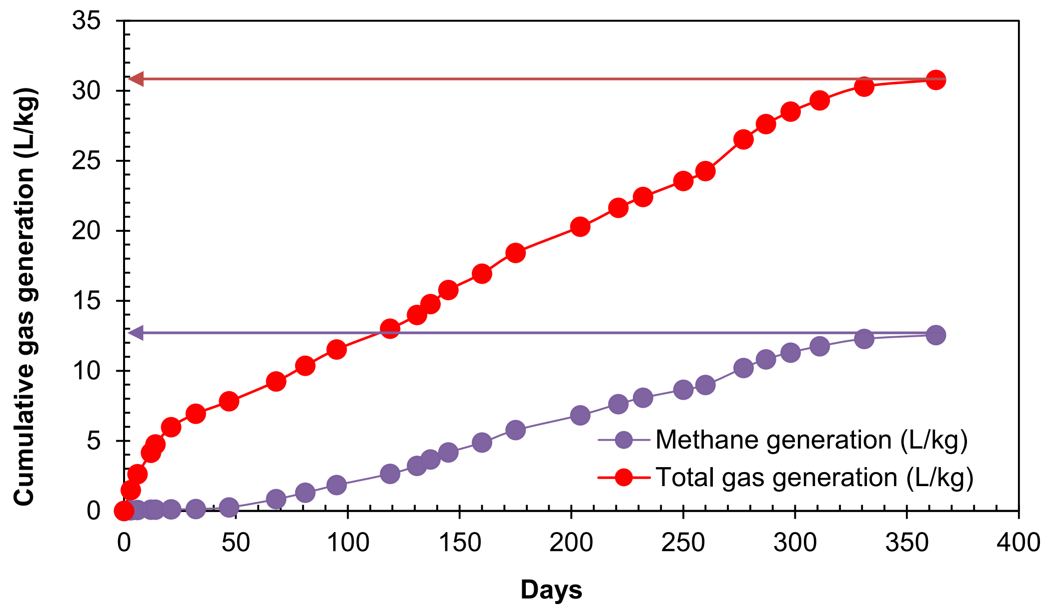


Figure 4-47 Cumulative total gas and methane gas generations in reactor R3 compacted at density 686.3 kg/m^3

4.8.2 Comparison of gas generation with the previous studies

In the literature, there was variation on the methane generation from laboratory bioreactor cells. These variation might be due to the difference in composition and amount of degradable materials present in waste. Leuschner et al., (1982) found the methane yield was 35 L/kg dry waste after 365 days from their experiments for a bioreactor operated with leachate recycle and buffer addition, and 63 L/kg dry waste for a bioreactor operated with leachate recycle and addition of buffer, nutrients and anaerobic digested sludge. Agdag et al., (2005) found the methane yield was 40 L/kg dry waste after 100 days from their experiments run with leachate recycle and buffer addition. Chiemchaisri et al., (2002) found the methane yield was 51.6 L/kg dry waste after 240 days from their anaerobic bioreactor operated with addition of buffer and anaerobic digested sludge. San et al. (2001) found the total methane produced was 34.4 L/kg dry waste after 275 days from their anaerobic bioreactor operated with leachate recycle.

Hao et al., 2008 performed decomposition of municipal solid waste (MSW) under oversaturated condition in comparison with leachate recirculation was investigated in two simulated reactors (A and B). The experiment was conducted in two identical reactors (A and B), constructed using a transparent, rigid, Plexiglas cylinder with a length of 150.0 cm and an inner diameter of 35.0 cm. Auxiliary equipment consisted of water/leachate recirculation system and gas measurement system. Altogether 39.0 kg of shredded synthetic MSW was filled into each of reactors with an average density of 325 kg/m^3. They operated reactor B with leachate recirculation while reactor A was maintained an oversaturated condition without leachate recirculation. Their results showed that MSW decomposition in reactor A was much faster than that in reactor B. At the end of experiment, the total biogas and methane yields were 2800 L and 1330 L in reactor A, while only 1470 L and 600 L in reactor B. They found the methane yield was 34.1 L/kg and 15.38 L/kg from reactor A and reactor B, respectively. In addition, daily biogas production was very sensitive to the fluctuation of ambient temperature and profoundly affected by the seasonal change of ambient temperature. The Figure 4-48 explains the production of methane and temperature variation.

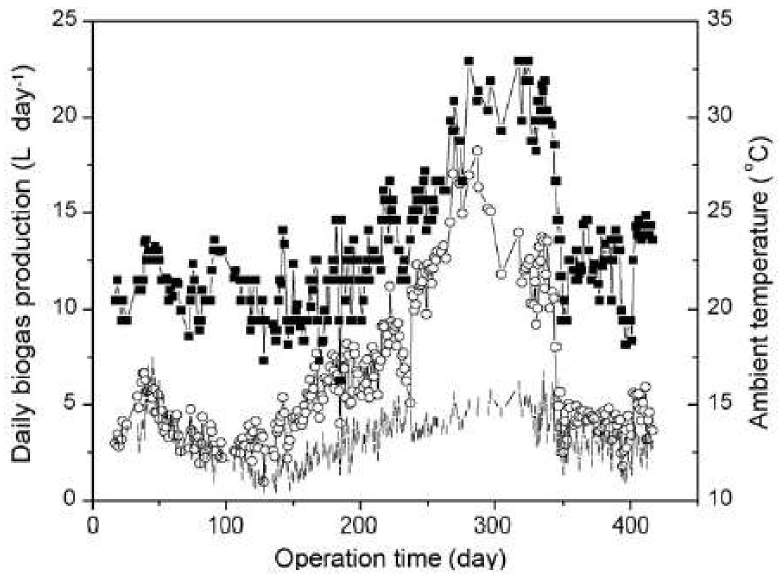


Figure 4-48 Variations of ambient temperature (&) and daily biogas production in reactor A (o) and reactor B (I) (Hao et al., 2008)

Al-Kaabi 2007 operated two groups of laboratory scale bioreactor cells in order to study the impact of saline water and sludge addition on the biodegradation of MSW. The first group (R1-R4) was operated without sludge addition. The second group (R5-R8) was operated with addition of sludge. The salt concentrations in the two groups were maintained at 0%, 0.5%, 1%, and 3% (w/v) respectively. All bioreactors were operated at neutral pH levels with leachate recycle. The author reported the total methane yield was 70.6, 61.7 and 47.5 L/kg dry waste for bioreactors R1, R2 and R4, respectively; and 84.7, 78.7, 72.6 and 59.0 L/kg dry waste for bioreactors R5, R6, R7 and R8, respectively. The methane generation for bioreactors R1, R2 and R3 are shown inFigure 4-49

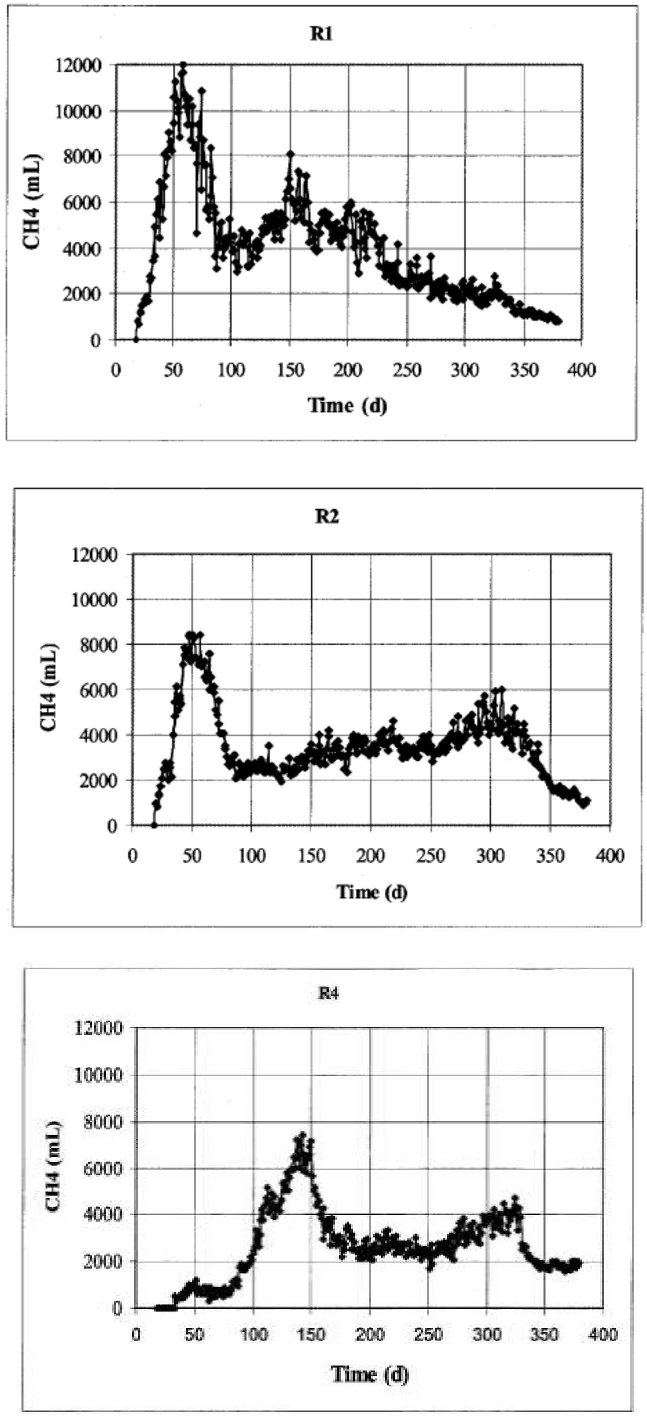


Figure 4-49 Daily methane production without using sludge (Al-Kaabi 2007)

Erses et al, (2008) observed cumulative methane generation as 158 L/kg dry solid waste which was high as compared to current research data. The main reason of getting high amount of methane was due to different in composition of waste. The authors taken synthetic solid waste mixture representing typical solid waste composition of Istanbul which consisted of 45% organic material (food + garden), 14.5% paper, 9.5% plastic, 5.6% textile, 3.8% glass, 2.2% metal, 4.4% ceramic, 15% other materials (dust, wood, brick, miscellaneous) by weight.

Sivanesan (2012) observed the two group of laboratory scale bioreactor cells in order to study the impact of saline water on the biodegradation of MSW. The author reported the total methane produced in reactors 1 and 3 with tap water were 325.82 L and 923.89 L. The methane gas generation rates were 39.8 L/lb (87.74 L/kg) and 100.5 L/lb (221.56 L/kg) dry weight of MSW as shown in Figure 4-50. For the reactor 4 with saline water, there was a lag period of 60 days prior to methane generation. The methane generation did not accelerate until 80 days. The cumulative methane generation of reactor 4 was 83.0 L/lb (182.98 L/kg) at the time of dismantling after 194 days, and reactor 4 was still producing gas.

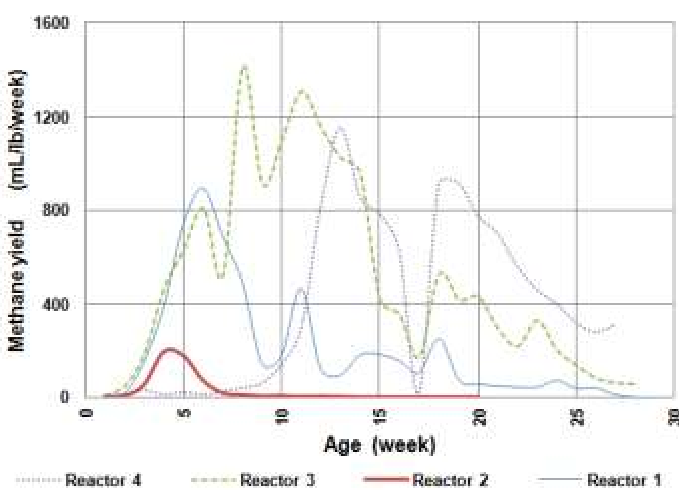


Figure 4-50 Rate of methane yield with time on weekly basis (Sivanesan 2012)

4.8.3 Variation on pH of leachate

The change in the pH level can be used as an indicator for the progress of the biodegradation process in the solid waste. It reflects the establishment of the acidogenic and methanogenic stages. The variation of pH profile of the leachate from different compacted samples over time is provided in Figure 4-52. Any kind of materials were not used in the MSW while in the compaction so that the time duration to change pH from acidic to basic was relatively long in the current research. Basically sludge was used by many researchers in order to accelerate the degradation process. Al-Kaabi (2007) reported Sludge addition was able to improve the methane yield. Using the sludge also reduces the initial adjustment time and transition of MSW from aerobic to anaerobic phases. It provides the friendly environment for the bacterial multiplication. The sole purpose of the research was to observe the effect of compaction on degradation of MSW without other's parameters influences. Neither of any materials were used in order to accelerate the degradation process. Leachate was the only material recirculated in order to accelerate the degradation process. The time of pH changing was less in the lowest compacted reactors and it was increased with the compaction level. The pH changing time from acidic to basic was around 110 days for the lowest compacted sample, 125 days for the medium compacted samples and 150 days for the highly compacted samples. During these times pH levels were approximately varying from 5.5 to 7.0 or on the acidic side of the pH scale. In most of the research the time duration from acidic to basic was lower than this current research. It was basically happened because of not using sludge in the reactors. Barlaz et al., 1990 reported the highest pH levels because of the sludge buffering capability while using the primary sludge in the bioreactor cell. During the intermediate anaerobic degradation stage, methanogenic bacteria slowly start to appear. As the methane gas production rate increases, hydrogen, carbon dioxide and volatile fatty acid concentrations decrease (Murphy et al., 1995). The conversion of fatty acids causes the pH within the waste cells to increase. The methane fermentation phase occurs when the pH levels for the two bioreactor cells rise to the neutral ranges of 7.0. Gas generation increased after pH increased to more than 7.0 in the two lower and medium compacted samples. But in the highly compacted sample there was not any type of correlation between pH and gas generation. The gas generation was not changing significantly even though pH was changed to alkaline state. There was not clear peak pH levels on all three reactors but after changing pH to alkaline state, it was almost showing the constant values for almost one year of bioreactor cell operation. Several studies in the literature reported pH values of leachate collected in the MSW landfills. Rees (1980) found that pH values of Aveley Landfill in the UK were between 7 and 8.5. Chu et al. (1994) studied the properties of leachate produced in two landfills in Hong Kong and the pH of the leachate to range of 7.2-8.0 and 7.2-8.4.

Warith et al., (1998) found that the initial pH of the leachate sample with buffer addition was 6 and it stabilized at approximately 7 after 25 days. Chiemcharis et al., (2002) found that the pH of leachate from anaerobic bioreactors operated with the addition of buffer and anaerobic sludge reached 7 after 100 days in their experiments. Warith (2002) observed that pH level was between 5.3 to 6 in first two weeks of reactor operation and 8.0. After 20 weeks, the pH level decreased to around 7.0 due to low methanogenic bacteria activity and starting of the maturation phase. The author reported the methane fermentation phase occurred when the pH levels for bioreactor cells rise to the neutral ranges of 6.8–8.0. This methane fermentation phase started between weeks four and five. The peak pH level for all bioreactors occurred on week six. After the initial pH increase, the pH levels slowly declined to a pH of about 7.

Rendra et al. 2003 found that the initial pH of leachate from their anaerobic bioreactor run with addition of buffer and anaerobic digested sludge was 5.7 and reached 7 after 112 days. Agdag et al. 2005 observed that the initial pH of leachate from a reactor operated with the addition of buffer was 5.3 and it reached approximately 7.0 after 20 days of bioreactor cell operation. Bilgili et al., 2007 reported the Influence of leachate recirculation on aerobic and anaerobic decomposition of solid wastes. According to their results, the pH values were in the range of 4–6 in the first 30 days of degradation in all reactors. After 30 days, pH values began to increase and reached to 8.0 after 100 days in aerobic reactors. After that, no considerable change was observed in pH of leachate from aerobic landfill reactors and measured between 8 and 9. Besides these, the pH values were 6.3 and 6.0 in anaerobic reactors around 100 days. On day 250, when aerobic landfilling operation was terminated, pH of the leachate from anaerobic (AN1 and AN2) reactors were 7.2 and 6.7, respectively. These results show that when aerobic degradation of solid wastes completed, the anaerobic reactor reaches to optimal pH values for anaerobic degradation, indicating the rapid degradation of solid wastes in aerobic conditions as shown in Figure 4-51. The results from this study are consistent with some results presented in the literature.

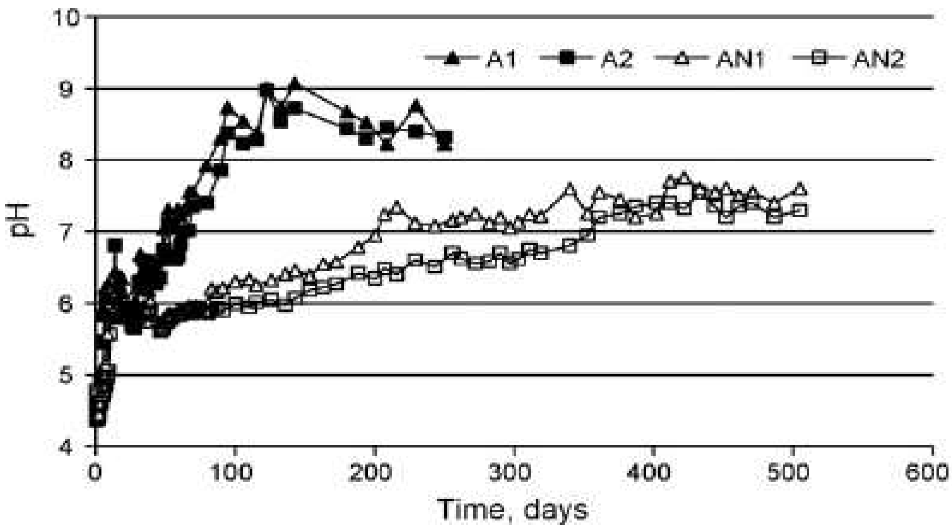


Figure 4-51 Variation of pH in aerobic and anaerobic landfill reactors with time (Bilgili et al., 2007)

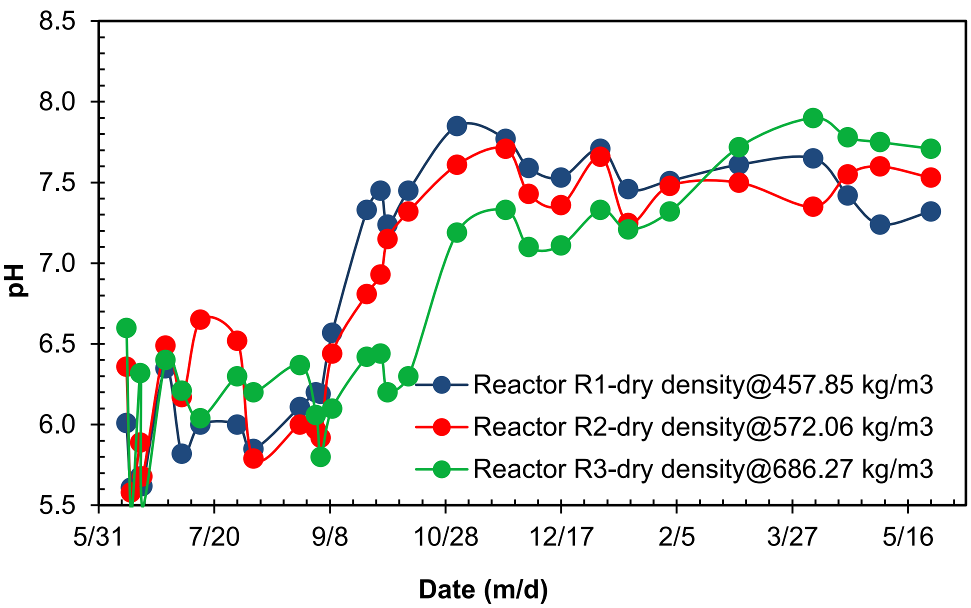


Figure 4-52 Variation of pH in anaerobic reactors R1, R2 and R3

4.8.4 Leachate recirculation

Leachate was recirculated through reactor R1, R2 and R3 in a similar way but the rate of leachate absorption was quite different among R1, R2 and R3 reactors. The difference was basically due to state of compaction level. In the beginning leachate was recirculated until leachate coming from the bioreactor cells. The initial moisture content of all waste samples were 40% on dry basis. In order to observe the sole effect of compaction, the composition of MSW kept exactly similar in all reactors. Samples were compacted uniformly using the tensile compression machine. In the beginning, water/leachate was circulated on 5% by weight on all samples and all reactors R1, R2 & R3 accepted all circulated water. After second time, R3 did not accept all the water when trying to circulate on the fix rate. Water/leachate was tried to circulate all time times in the ratio 1:1.25:1.5 on reactor R1, R2 and R3, respectively. The reason of doing the circulation was based on weight of MSW. The weight of MSW was taken in the ratio of 1:1.25:1.5 on reactors R1, R2 and R3, respectively. Reactors R1 and R2 accepted all water/leachate when trying to circulate at 5% by weight i.e. 1 liter on Rectors R1, 1.25 L in R2. The circulation was fast on the reactor R1 and it accepted all liquid easily all the times. After saturating the waste samples, the liquid acceptance capacity of R2 also decreased and did not accept all water on most of the times. That was the reason water was circulated more in R1 even though water/leachate was tried to circulate on 1:1.25:1.5 ratio on reactors R1, R2 and R3, respectively. As the void space is greatly affected by the density, the void space available of reactor R1 might be high as compared to other two reactors R2 and R3 so that the R1 did not take much time while performing circulation. The total circulation of water/leachate is explained in the Figure 4-53.The total circulation of water/leachate was 28L in reactor R1, 20.5L in reactor R2 and 6.5 L in reactor R3, respectively.

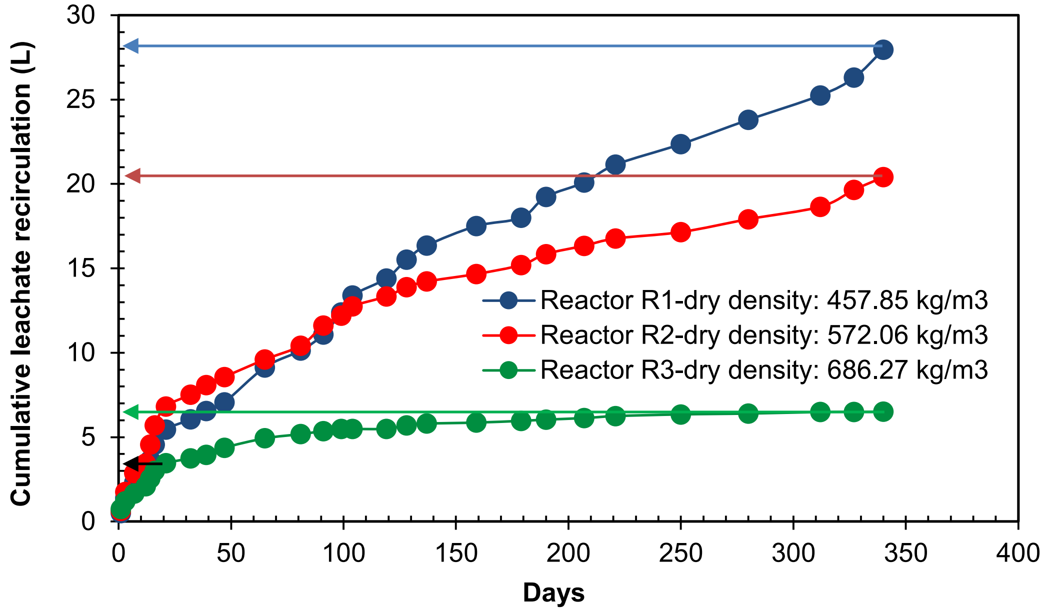


Figure 4-53 Cumulative leachate recirculation in anaerobic reactors R1, R2 and R3

4.8.5 Leachate generation

The leachate generation through reactors R1, R2 and R3 was monitored. As the circulation was different in three reactors, there was quite different on the generation of leachate. The difference was basically due leachate acceptance by the reactors while performing circulation. In the beginning, the leachate generations through reactors R1 and R2 were similar but the generation through reactor R3 was always in small amount. After 60 days, there was complete variation on the generation on the reactors R1 and R2 also. The total leachate generations through reactors R1, R2 and R3 were 25 L, 15 L and 3 L in reactor R3, respectively which are shown in Figure 4-54.

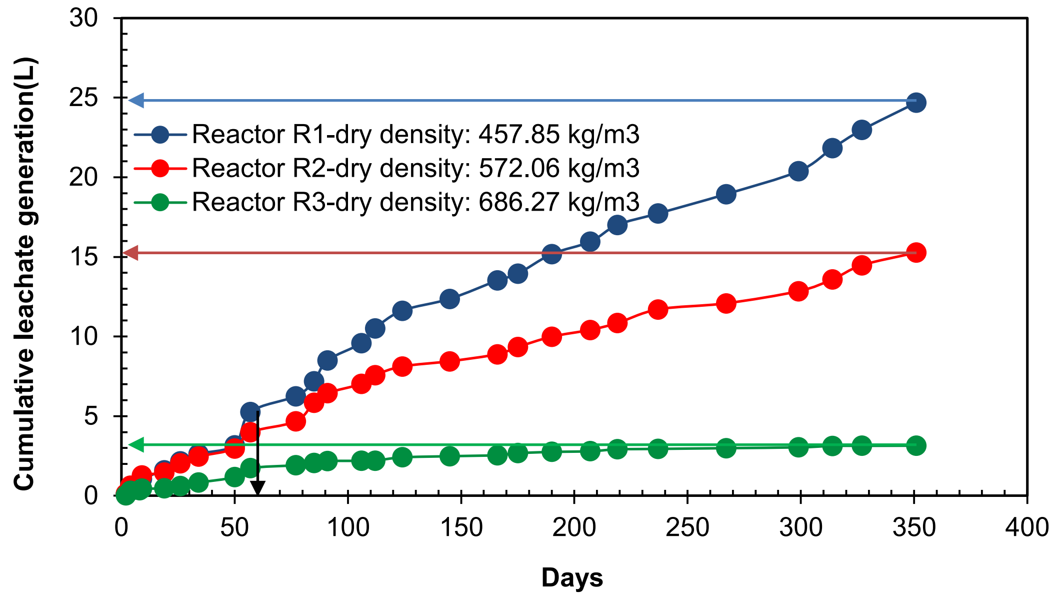


Figure 4-54 Cumulative leachate generation in anaerobic reactors R1, R2 and R3

4.8.6 Variation of hydraulic conductivity

Constant head permeability tests were performed once in a month on all reactors R1, R2, and R3 along with collecting generated gas. The cells are equipped also with leachate collection, leachate recirculation, gas collection along with capability of performing permeability tests which are shown in previous chapter Figure 3-24. Hydraulic conductivities were measured on the monthly basis to observe the effect of degradation on the flow behavior. Hydraulic conductivity tests were performed by removing all the entrapped air that was generated due to degradation process. The entrapped air was always removed by applying water from the bottom of the samples. This process removes almost all generated gas that was stored in void space inside the samples. The process is explained in previous chapter 3 in Figure 3-11. The variation of the hydraulic conductivities at different dry densities for the waste is shown in Figure 4-55**Error! Reference source not found.**. In the beginning the coefficients of hydraulic conductivities decreased in R1 and R2 up to 3 months and the values not changing significantly around 8 moths. The trend of hydraulic conductivities curves were almost similar for the reactors R1 and R2. It can be concluded that the degradation had no any effect on the hydraulic conductivities values in reactors R1 and R2 compacted samples but increased slightly the final values as compared to initial values. Whereas the hydraulic conductivities were continuously decreasing in reactor R3 sample which indicated that there might be gas trapped inside the waste because the degradation level of reactor R3 was very low. The degradation level in reactors R1 and R2 were quite high as compared to reactor R3. The gas generation was quite high in reactors R1 and R2 whereas the gas generation in reactor R3 was almost negligible as compared to R1 and R2. Even though there was high degradation of solid waste in reactors R1 and R2 but the hydraulic conductivities were not varying but slightly increased after 8 months. There was not much degradation on the reactor R3 sample but the hydraulic conductivity decreased every month which indicated that hydraulic conductivity might decrease due to accumulation of gas rather than degradation. The coefficients of permeability varies from 8.72×10^{-4} cm/sec to 4.01×10^{-4} cm/sec, 2.73×10^{-4} cm/sec to 1.56×10^{-4} cm/sec and 3.78×10^{-5} cm/sec to 2.27×10^{-7} cm/sec for R1, R2 and R3 reactors.

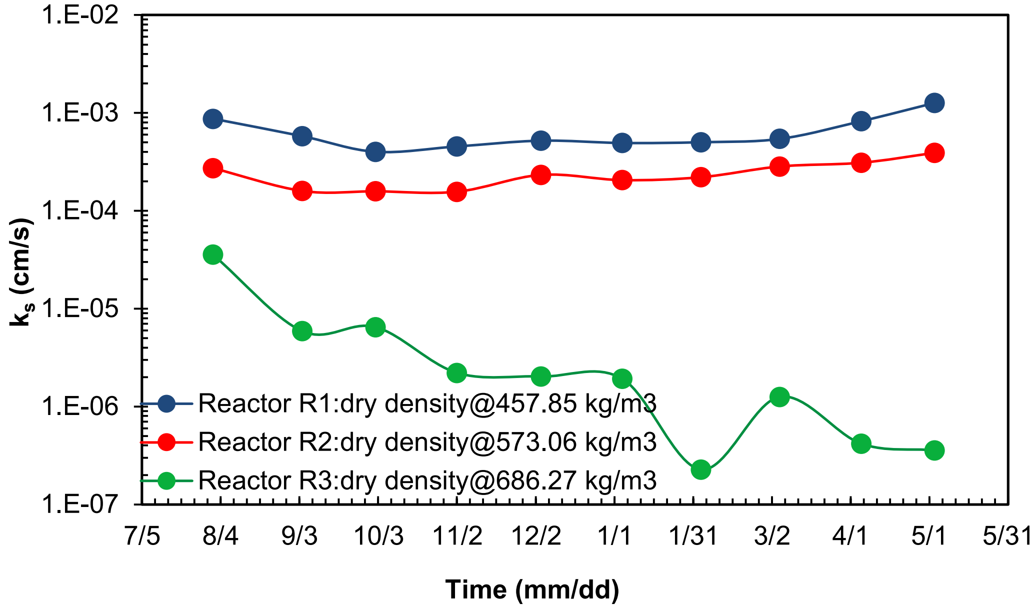


Figure 4-55 Variation of hydraulic conductivity on R1, R2 and R3 reactors

4.8.7 Moisture variation on waste samples

The moisture content of the MSW samples were maintained at 40% by dry basis in order to make similarities among the samples. Initially the composition of each samples were same and moisture contents were also kept same in order to observe the effect of compaction on the degradation of MSW. It was also reasonable to observe the effect of compaction on the final level moisture content of MSW due to the effect of degradation. Generally moisture content was determined before putting the waste into the simulated bioreactor landfills (initial) and at the end of the study for three reactor samples (final) by drying approximately 2 lb. of degraded samples. The final moisture content of the bioreactor samples were 99.0% on lowest density sample, 72.2% on medium density sample and 51.9% on highest compacted samples on the dry basis, respectively. The moisture content of MSW was increased with the degree of decomposition. Lowest density has higher porosity so that it could hold more moisture than other compacted samples. As the density is the main factor affecting porosity, it is not reliable to conclude degradation effect on moisture holding capacity of the waste. The initial and final moisture content results of reactor samples are presented in Table 4-25.

Table 4-25 Variation of moisture content of compacted samples

Sample Density (kg/m3)	Initial Moisture content (%)		Final Moisture content (%)	
	Dry basis	Wet basis	Dry basis	Wet basis
457.8	40.0	28.57	99.0	49.8
572.1	40.0	28.57	72.2	41.9
686.3	40.0	28.57	51.9	34.2

4.8.8 Visual inspection of degradation

This is also a method to estimate the label of degradation of MSW. Generally, the waste was very clear and had no any black color in fresh state. As the degradation occurred the color changed into black which was observed from outside because of the transparent materials used in reactors. The degraded waste seemed blacker as compared to less degraded waste. The gas generation data indicated that reactor R1 was more degraded as compared to R2 and R3. Reactor R3 was less degraded among all bioreactors. This is one of the simplest method of determining the status of degradation but it could not give exact idea about methane potential of waste, level of cellulose, hemicellulose, lignin and label of volatile solid. The visual inspection was generally followed in sampling of degraded waste and in boring in landfill. Besides, the methane gas generation from waste, the (C+H)/L ratio is also one of the true indicator of status of degradation but the current research had not focused on the (C+H)/L to estimate the degradation status.

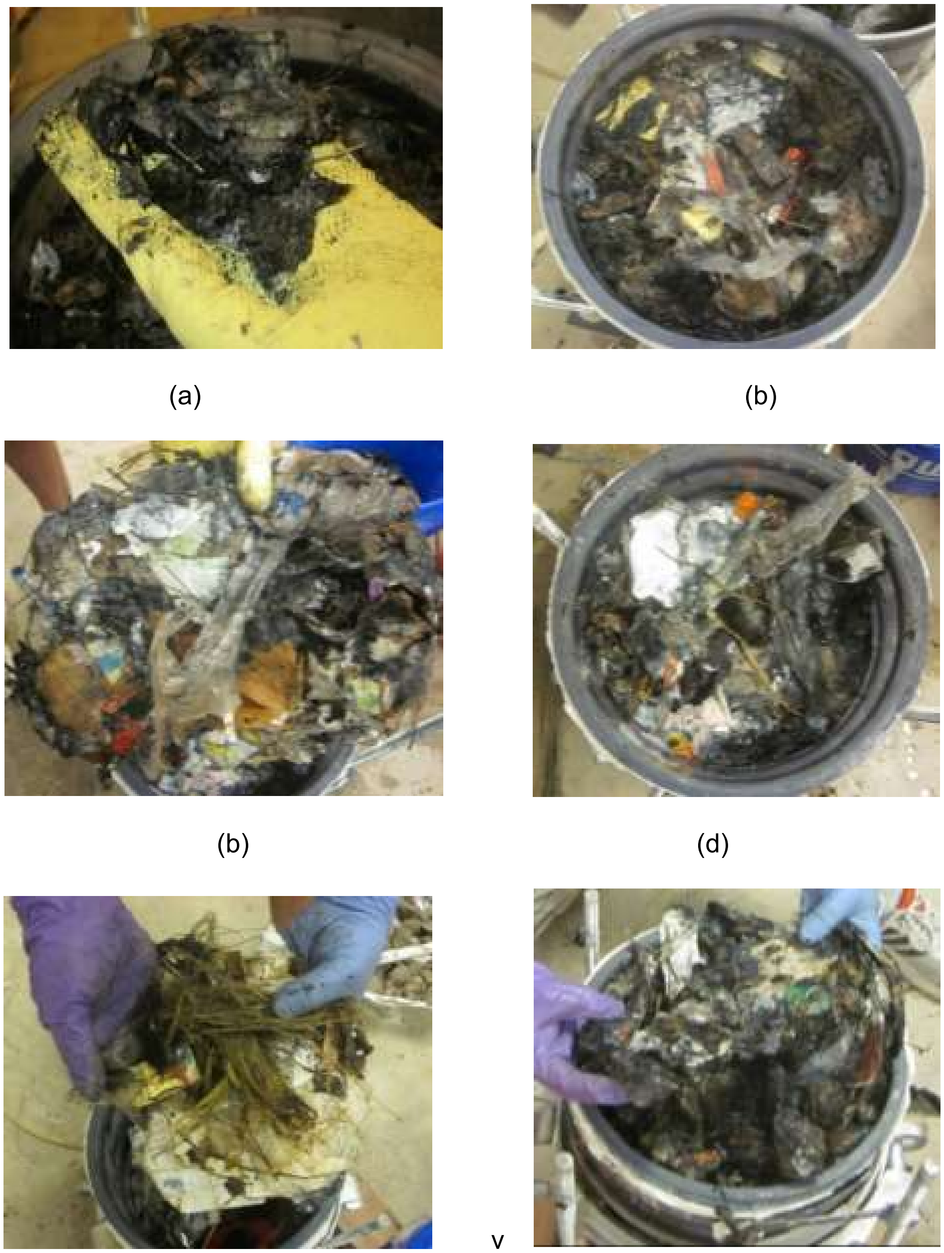
As the degradation started, there was change in color of the waste which was observed from outside. The reactors were made of transparent PVC pipe which enabled to visualize the inside waste from outside. The indication of color of the waste matched to the degradation of the waste. This is just a method for comparison among degraded waste in the reactors because it cannot provide any exact data. It was very clear the reactors R1 and R2 had darker color waste than reactor R3 which was shown in Figure 4-56**.**

When the reactors were dismantled, it was surprising to see the fresh color of reactor R3 which was the indication of less or not degradation. The color of waste in R3 was very clear and clean in deeper part of reactor. The reactors R1 and R2 had waste which were darker and seemed highly degraded. The color of waste in the same reactors were also varied with the depth. The top part of the waste was blacker than the bottom and this was more distinctly observed in reactor R3. The Figure 4-57 clearly explained the degradation status indicated by color.

(a) (b)

(c)

Figure 4-56 Comparison of degradation with variation of color of degraded MSW in (a) R1 (458.5 kg/m^3), (b) R2 (572.1 kg/m^3) and (c) R3(686.3 kg/m^3) reactors



(a) (b)

(b) (d)

(d) v (e)

Figure 4-57 Visual inspection of degradation (a) & (b) reactor (R1) 457.8 kg/m^3, (b) & (c) reactor (R2) 572.1 kg/m^3, (d) & (e) reactor (R3) 686.3 kg/m^3

4.8.9 Variation of volatile solids

Volatile solids (VS) of degraded waste were determined to study the effect of compaction on the degradation. The volatile solids content of degraded waste and fresh waste are tabulated in Table 4.26. Further, the change in the volatile solids after burning in muffle furnace is shown in Figure 3-19 . As previously discussed, the volatile solids test also provide a good indication of degradation level of waste mass or amount of biodegradable material that is remaining in the waste mass. The waste considered here was mixed of all components present in waste. All three samples had the initially similar composition so that the volatile solid was considered same in all three reactors. In a mixed solid waste sample from a landfill, some portion of the waste is comprised of inorganics such as soil, plastic, Styrofoam, sponges, C&D debris and metal. Over time, as the degradation increases, the amount of organic waste decreases, while the inorganic portion remains same. The change in volatile solids in R1, R2 and R3 reactors as compared to fresh waste were found to be 9.96%, 9.00% and 6.35%even after operating the reactors for a period of 12 months. The reduction in volatile solids were 13.01%, 11.76%, 8.30% for the reactors R1, R2 and R3, respectively. The reduction in volatile solids was very low in the current study as compared to previous studies.

Several studies have been conducted to determine the amount of volatile solids during the stages of biodegradation of MSW. Haque (2007) reported that the average initial (phase I) volatile solid percentage was 91.5 % and it decreased to 46 % in phase 4. Al-Kaabi et al. (2009) found that recirculating leachate with salinity level of 0%, 0.5%, 1.0%, and 3% reduced volatile solids by 84%, 78%, 74%, and 66%, respectively, at the end of the anaerobic stage. Sivanesan (2012) found the reduction in volatile solids were 84%, 82%, 77%, and 70%, respectively, when the sludge was added for the same salinity values in the re-circulated leachate. The average percentage reduction in volatile solids of 75.7% increased to 78% when the sludge was added to the reactors.

Table 4-26 Comparison of volatile solid in fresh and degraded MSW

Reactors-Density	Initial VS (%)	Final VS (%)	Change in VS
R1-457.8 kg/m^3	76.54	66.58	9.96(%)
R2-572.1 kg/m^3	76.54	67.54	9.00
R3-686.3 kg/m^3	76.54	70.19	6.35

Chapter 5

Estimation of optimum compaction

5.1 Introduction

It was most important to understand the effect of compaction on the properties of the municipal solid waste and determine the optimum compaction for successful bioreactor operations. The flow pattern within the landfill waste will widely vary with different factors such as anisotropy, heterogeneity, partial saturation, presence of landfill gas, variations in waste density and effective overburden stresses within the landfill (Beaven et al., 2011). Previous studies had not focused on the determination of the optimum compaction level for the bioreactor landfill operation. A certain compaction level might be good for bioreactor landfill operation which can provide uniform flow and can have sufficient drainage capacity. Besides flow criteria, certain compaction level might be good for the microbial activity within the landfill and can produce maximum methane gas. Drainage capacity of the solid waste is an important parameter for the bioreactor landfill which is required to maintain uniform flow and producing sufficient leachate from waste degradation. In this chapter, the sole focus was to estimate the compaction level based on flow parameters, such hydraulic conductivity and porosity and methane gas generation from the solid waste. Maximum ranges in densities were considered in the current studies. The purpose of the selection of large ranges in densities was to observe the behavior of flow and variation in porosities of the wastes. Different methods were applied to estimate a range of compaction level for MSW. After certain intervals, there were changes in the nature of the curves of hydraulic conductivities and porosities of the wastes. The tangent intersection method was used to estimate the changing point, which was considered the optimum density for the municipal solid waste in order to maintain the required flow.

5.2 Optimum compaction for MSW

Optimum density for waste was estimated from considering different hydraulic conductivity, retained/effective porosity, drainable porosity and degradation of MSW. Different perspectives were utilized to conclude the optimum compaction level. In this study, fresh waste was only taken to conduct tests. Since the work is related to compaction required for the waste, several fresh waste with the exactly similar composition were tested. Basically the compaction level was determined by considering the mainly 3 parameters in this current study. The optimum density might depend on many factors, the optimum density was determined by considering the hydraulic conductivity, porosity and degradation of fresh waste. To estimate the optimum density for fresh waste, three type of wastes for hydraulic conductivity and porosity and one type of waste for degradation were selected and performed tests. The factors which can affect hydraulic properties such as permeability, porosity are already discussed in previous chapter. The factors are not considered while determining the optimum density.

5.2.1 Optimum density from hydraulic conductivity

The concept of operating the landfill as a bioreactor emerged from the addition of moisture that stimulates microbial activity by providing better contact between the waste and microorganism via solvent medium. One of the most important parameter to maintain the flow and thus to operate bioreactor landfill successfully is the hydraulic conductivity of waste. It governs the flow and thus balances the moisture distribution inside the landfill. If the density of municipal solid waste is too high it can creates the dry condition and cannot transport liquid from one part to another part. If the moisture cannot circulate along the landfill, waste cannot degrade and remain dry for long period of time which is not beneficial for bioreactor landfill. The most common technique to accelerate degradation process is to stimulate microbial activity by adding moisture to the waste. In order to transport liquid to landfill hydraulic conductivity is the vital parameter. If the hydraulic conductivity is more, the liquid can easily flow. If the liquid can reach to all the place, that will accelerate degradation process by creating friendly environment for the microorganisms. It is obvious that hydraulic conductivity is going to be higher if the density is lower. Several previous studies showed the relationship between density and hydraulic conductivity of waste. It might be better to have higher hydraulic conductivity for waste but it might create waste stability and degradation problems. If the density is too low, waste materials can absorb huge amount of liquid and waste might float over the liquid. If flooding condition occur due to more water, this is the most critical condition and can create problem for health and safety. Also if there is more water present than required, that might not be beneficial for the microbial activity. In this study, it has been focused heavily on hydraulic conductivity with and without time effect to come up with optimum range of density.

5.2.1.1 Hydraulic conductivity without consideration of time

A number of series of saturated hydraulic conductivity tests were carried out on various fresh MSW such as shredded waste-A, shredded waste-B and un-shredded waste-B. The tests for shredded waste were carried out in three different size small, medium and large size diameter permeameters while tests for unshredded were carried out in large permeameter.

The hydraulic conductivity against dry density was plotted for all wastes. The graphs of hydraulic conductivity versus dry density, at each density obtained from small; medium and large size rigid wall permeameters. The results clearly demonstrated that the hydraulic conductivity of MSW can be significantly influenced by increased density. There were very different values on the hydraulic conductivity in the same density in previous studies. The main reason of difference might be composition, unsaturation state due to air accumulation, mixing process of waste components, degradation level and amount of fine content and soil. In the current studies, composition is maintained exactly same for each samples. These results showed a definite correlation between dry density and hydraulic conductivity. While plotting graph between hydraulic conductivity versus dry density of waste, similar trends were observed for all samples, utilizing all permeameters. Hydraulic conductivity decreased with increasing density, but the nature of the curve changed abruptly at a certain density point. A tangent intersection method was used to find out the cutoff point in the hydraulic conductivity curve. Two straight tangent lines were drawn from the initial point and the final point. Two tangent lines were extended to reveal the cutting point. A vertical line and a horizontal line were drawn from the cutting point towards the horizontal density axis and vertical hydraulic conductivity axis. The values of densities obtained from the hydraulic conductivity curves for all permeameters were compared. It was interesting to observe almost the similar densities in all cases. A horizontal line was drawn on vertical axis to determine the value of hydraulic conductivity based on optimum density. The value of hydraulic conductivity was determined from the vertical axis can be defined as minimum hydraulic conductivity to maintain the required flow.

The permeability of the shredded waste-A varying from 2.76E-02 cm/s to 2.60E-06 cm/s for the dry density between 347.7 kg/m^3 and 714.3 kg/m^3. It also followed a curve path. It had shown a cutoff point while drawing tangent lines on the left and right side of curve. The optimum density was obtained 512 kg/m^3 through the small permeameter. A horizontal line was drawn on vertical axis to determine the value of hydraulic conductivity based on optimum density which has the value of 1.2E-03 cm/s. In the same way, the technique was applied for similar waste-A to get hydraulic conductivity curve through medium permeameter. Waste-A had the value of permeability ranging from

5.38E-03 cm/s to 3.09E-06 cm/s for the dry density between 324.2 kg/m^3 and 733.8 kg/m^3. Nearly similar nature of curve was obtained through the hydraulic conductivity versus dry density. The optimum density was obtained 520 kg/m^3 through the medium permeameter. A horizontal line was drawn on vertical axis to determine the value of hydraulic conductivity based on optimum density which had the value of 9.0E-04 cm/s.. The large 25.4 cm diameter permeameter resulted in a range of hydraulic conductivity of 4.96E-03 cm/s to 9.87E-06 cm/s for the dry density between 332.1kg/m^3 and 722.1 kg/m^3. The optimum density was obtained 510 kg/m^3 through the large permeameter. A horizontal line was drawn on vertical axis to determine the value of hydraulic conductivity based on optimum density which has the value of 1.10E-03 cm/s The estimation of optimum density for waste-A by utilizing different permeameters and minimum hydraulic conductivity corresponding to optimum density are summarized in Figure 5-1.

Similarly optimum density was determined from the hydraulic conductivity against dry density for the waste-B sample. All sample had the exactly similar composition with similar kind of individual components but particles size were different. The shredded waste-B had the value of permeability ranging from 1.29E-02 cm/s to 1.64E-06 cm/s for the dry density between 319.5 kg/m^3 and 730.2 kg/m^3 while using a small size permeameter. The cutoff point on the hydraulic conductivity curve was assumed as the optimum density which had the value as 482 kg/m^3 through the small permeameter. A horizontal line was drawn on vertical axis to determine the value of hydraulic conductivity based on optimum density, which had the value of 1.10E-03 cm/s. Waste-B had the value of permeability ranging from 3.69E-03 cm/s to 2.22E-06 cm/s for the dry density between 342.0 kg/m^3 and 734.0 kg/m^3 through a medium size permeameter. The same procedure applied and obtained the optimum density as 511 kg/m^3 through medium permeameter. A horizontal line on vertical axis gave the value of hydraulic conductivity 7.0E-04 cm/s for corresponding optimum density. The large 25.4 cm diameter permeameter resulted in a range of hydraulic conductivity of 4.28E-03 cm/s to 9.54E-06 cm/s and it has been observed similar type of curve between hydraulic conductivity and dry density. The optimum density was obtained 500 kg/m^3 through the large permeameter. A straight horizontal line on vertical axis gave the value of hydraulic conductivity 1.0E-03 cm/s for corresponding optimum density through this equipment. The estimation of optimum density for waste-B by utilizing different permeameters and minimum hydraulic conductivity corresponding to optimum density are summarized in Figure 5-2.

In the same way, optimum density is also determined for the unshredded waste-B. Exact similar procedure was applied to estimate the optimum density. The unshredded waste resulted the coefficient of permeability from 4.45E-03 cm/s to 9.65E-07 cm/s for the dry density 315.8 kg/m^3 and 720.9 kg/m^3. While applying the tangent intersection method to estimate optimum density, it gave the value of the optimum density as 490 kg/m^3 through the large permeameter. A horizontal line was on vertical axis gave the hydraulic conductivity of 9.0E-04 cm/s for corresponding to optimum density obtained through this equipment. The estimation of optimum density for waste-B by utilizing different permeameters and minimum hydraulic conductivity corresponding to optimum density are summarized in Figure 5-3.

While observing all optimum density for different fresh waste using different size permeameter, the range of optimum density was from 482 kg/m^3 to 520 kg/m^3 which was very close to each other. There were some discrepancy in the values which might be due to uneven density inside the compaction. It was one of the difficult process to get uniform density inside the permeameter for waste materials. Waste particle was so heterogeneous which had a huge difference in particle size although the waste was shredded. The heterogeneity created non uniformity on mixing of waste particles. It can be assumed that the range of optimum densities were very close. It was very obvious that the optimum range of density should be from 480 kg/m^3 to 520 kg/m^3 while considering the hydraulic conductivity. The average optimum density is around 500 kg/m^3. The hydraulic conductivities of waste at corresponding optimum densities were very close to each other for all waste through all permeameters. It can be assumed these densities were nearly the same and hydraulic conductivities were also same for optimum density obtained from all permeameters. Also the minimum range of hydraulic conductivity is varied from 1.0E-04 to 7.0E-04 cm/s. The optimum densities and minimum hydraulic conductivity for different waste samples are summarized in Table 5-1 and Table 5-2, respectively.

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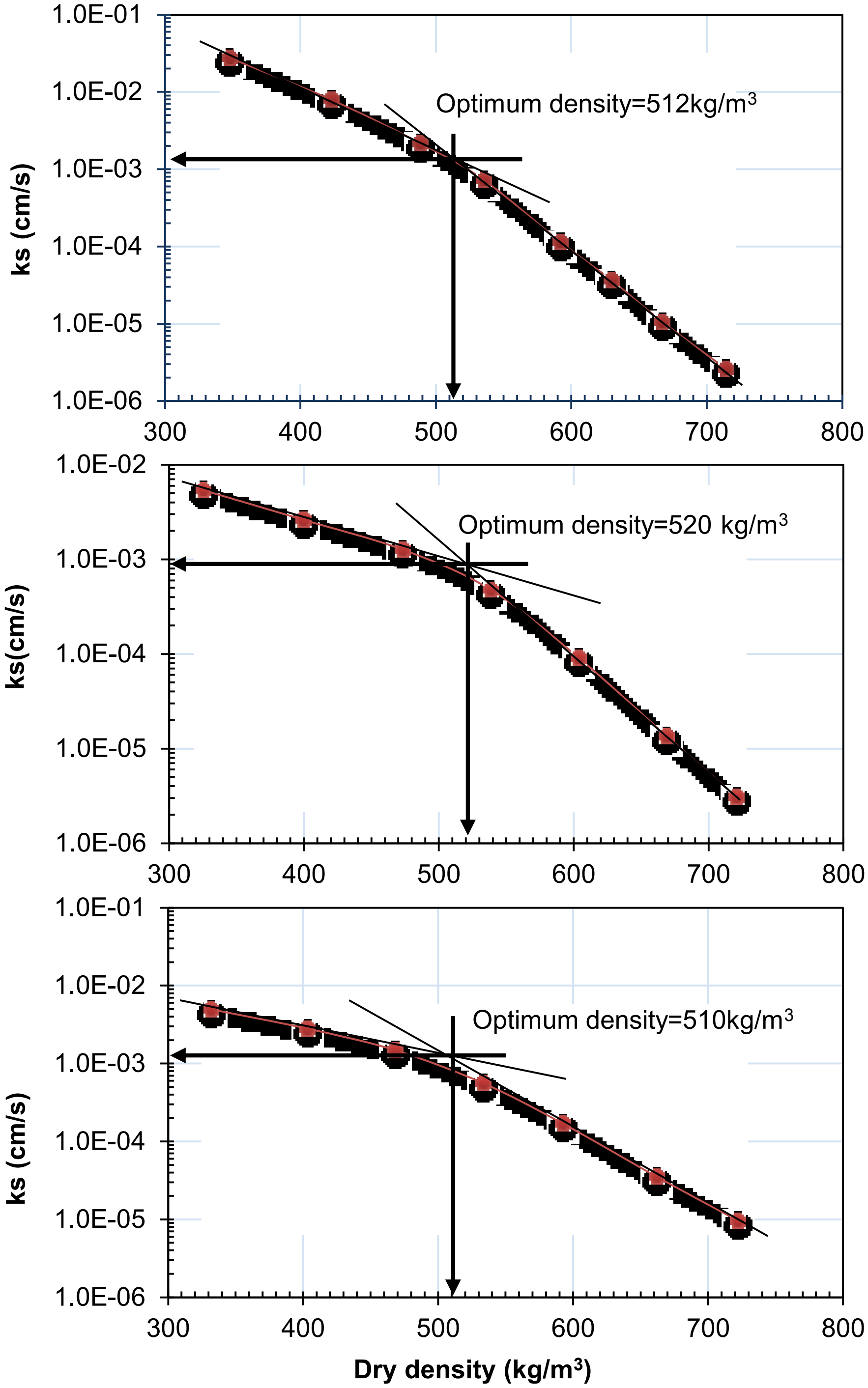

Figure 5-1 Estimation of optimum dry density from permeability for shredded waste-A data from 6.35 cm, 15.25 cm and 25.4 cm diameter permeameter

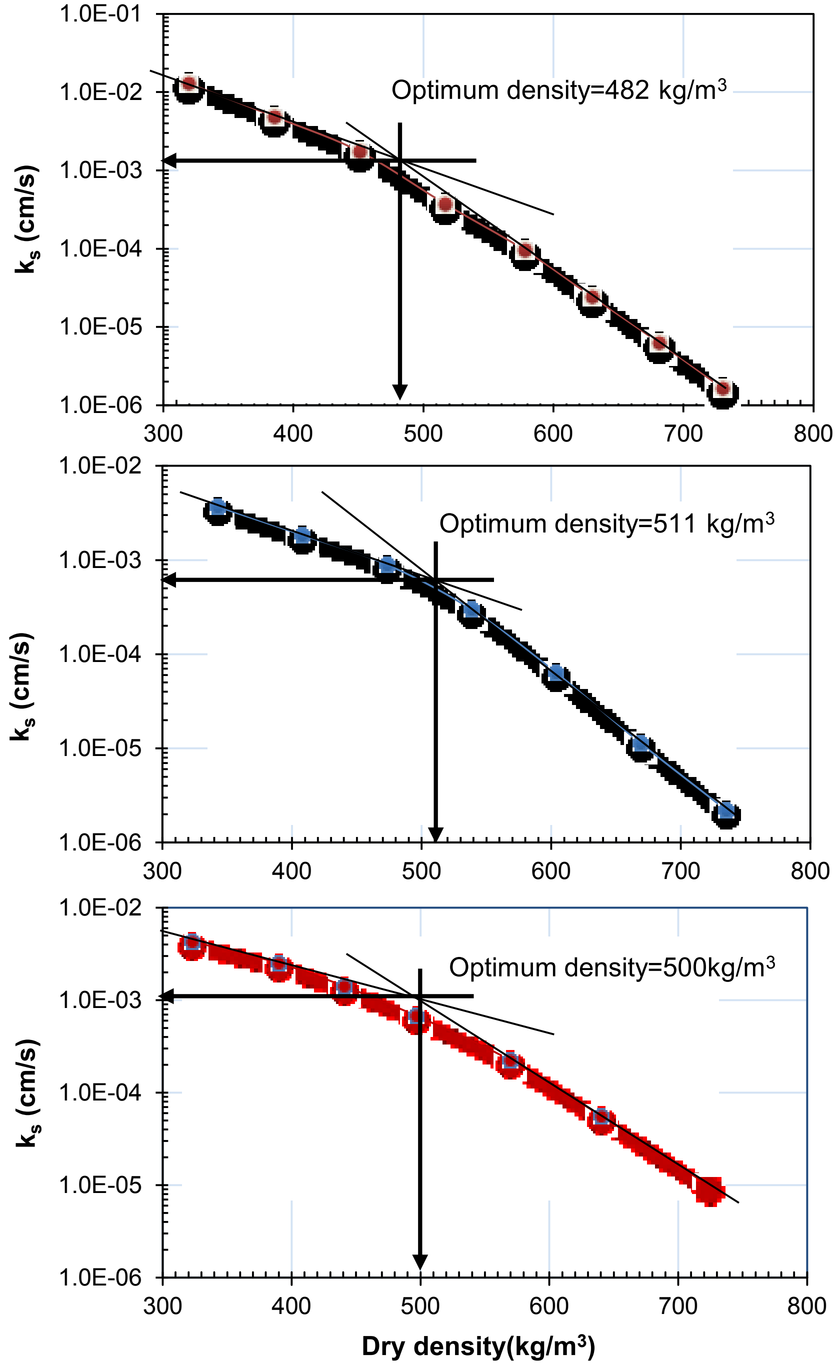

Figure 5-2 Estimation of optimum dry density from permeability for shredded waste-B using 6.35 cm, 15.25 cm and 25.4 cm diameter permeameter

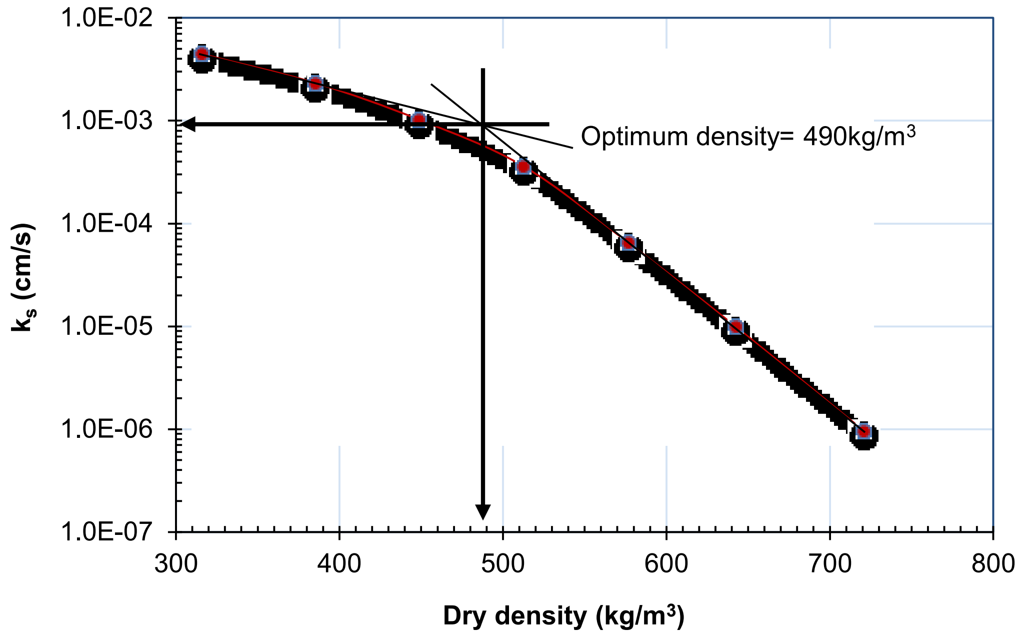

Figure 5-3 Estimation of optimum dry density for unshredded waste-B data from permeability using large 25.4 cm diameter permeameter

Table 5-1 Summary of optimum densities from hydraulic conductivity test results

Waste type	Optimum density (kg/m^3)		
	From small device	From medium device	From large device
Shredded waste-A	512	520	510
Shredded waste-B	482	511	500
Unshredded waste-B	NA	NA	490

Table 5-2 Summary of minimum hydraulic conductivity based on optimum density

Waste type	Minimum hydraulic conductivity (cm/s)		
	From small device	From medium device	From large device
Shredded waste-A	1.20 E-03	9.00 E-04	1.10 E-03
Shredded waste-B	1.10 E-03	7.00 E-04	1.00 E-03
Unshredded waste-B	NA	NA	9.00 E-04

5.2.1.2 Variation of hydraulic conductivity with time

The long term variation in hydraulic conductivity of waste-A was measured for 116 days. The purpose of the test was to observe the fluctuation on hydraulic conductivity over time at different compaction level. Although it might not be reliable to conclude optimum density based on results of only 3 density levels but it give a tentative range of density to maintain the flow. It is obvious that if there were many tests on variable densities, it would be more precise for the estimation of optimum density based on time factor. It is been already proposed by many researchers density will affect largely on the hydraulic conductivity. But there were not much studies available on the hydraulic conductivity variation with time. It is also one of the most important parameter need to be considered while designing bioreactor landfill. It is important to understand how the hydraulic conductivity varies on the landfill with respect to time. In this study time factor also considered along with density to estimate the range of required optimum density in order to maintain uniform flow for the landfill. It was observed hydraulic conductivity highly fluctuated over time for compacted waste-A samples but there was a unique pattern on the reduction of hydraulic conductivity. There were frequent reduction on the flow in the highly densified sample as compared to less compacted samples. If the flow was too small, air vent was opened and air was removed which increased the flow again. If the sample is not fully saturated, the hydraulic conductivity becomes very low due to suction. The ratio of reduction in hydraulic conductivity was more for the lowest and less for the highly compacted waste. Actually, this ratio was small in highly compacted sample because of frequently removing generated gas from the permeameter. When the gas was released the hydraulic conductivity increased again. It was also observed when the gas was released by applying water from the bottom of the sample, the hydraulic conductivity almost increased to initial value which was more pronounced in less densified sample.

This concludes that the density should not be more than 569 kg/m^3 while considering the long term flow. If the density is more than 569 kg/m^3 it might create problem to maintain the flow inside the landfill. As the landfill is very big so that the effect of gas generation will be more in actual landfill than in small sample in the lab.

The lowest densified sample is more porous so that more gas can be accumulated inside it as compared to highly compacted samples. Due to this reason, hydraulic conductivity was reduced quite frequently on highly compacted sample if the gas was not removed. The removal of gas on less compacted samples was less frequent as compared to highly compacted waste. Basically gas yield created the unsaturated condition and affect the flow. When the permeability tests were performed by removing the entrapped air by applying water through the bottom of sample the flow was increased significantly which was also supported by the results of hydraulic conductivity tests on waste-B samples on monthly basis. In waste-B, hydraulic conductivity tests were performed by removing all the generated gas through the application of water from the bottom of samples.

Similarly, constant head permeability tests were performed on the monthly basis to see the effect of degradation on the hydraulic conductivity. In this case, hydraulic conductivity tests were performed on waste-B by removing all the entrapped air that was generated due to degradation process. The generated air was removed by applying water from the bottom of the sample unless air bubble was observed.

The hydraulic conductivity was decreased significantly in case of 686.3 kg/m^3 dry density sample. It indicated that there should be gas trapped inside the waste which was almost impossible to remove all the gas. It was also observed that degradation was higher in 457.8 kg/m^3 and 573.1 kg/m^3 waste samples than highly compacted 686.3 kg/m^3 sample. Even though there was high degradation of solid waste in lower unit samples but the hydraulic conductivity was not changing much. There was not much degradation on the high unit weight sample but the hydraulic conductivity decreasing in every month which indicated that hydraulic conductivity might decrease due to accumulation of gas rather than degradation. In spite of decreasing, the hydraulic conductivity increased slightly from 9.5E-04 cm/s to 1.4E-03 cm/s over the time in less densified (457.8 kg/m^3) sample. Similarly the hydraulic conductivity increased slightly from 2.8E-04 cm/s to 4E-04 cm/s over the time in another compacted (573.1 kg/m^3) sample. While the hydraulic conductivity decreased from 3.5E-05 cm/s to 4.0E-07 cm/s over the time in highly compacted (457.8 kg/m^3) sample.

The permeability reduction ratio on two compacted samples were 3 and 2.86 on 457.8 kg/m^3 and 573.1 kg/m^3, respectively. The reduction in permeability on three compacted waste samples are analyzed in Figure 5-4. These two samples were more degraded samples which were also supported by the huge gas generation from these samples. The permeability reduced slightly in these two samples although high degradation occurred. Whereas, the permeability reduction ratio on most compacted samples was 175 though less degradation occurred. This concluded that, if the degradation within these range, flow will not be that much affected due to degradation on time. It can be observed very clearly, the density should be less than 573.1 kg/m^3 while considering the long term flow for the bioreactor landfill. It is clear that if the density is below this value, the flow can be maintained by supplying of water and releasing the generated landfill gas.

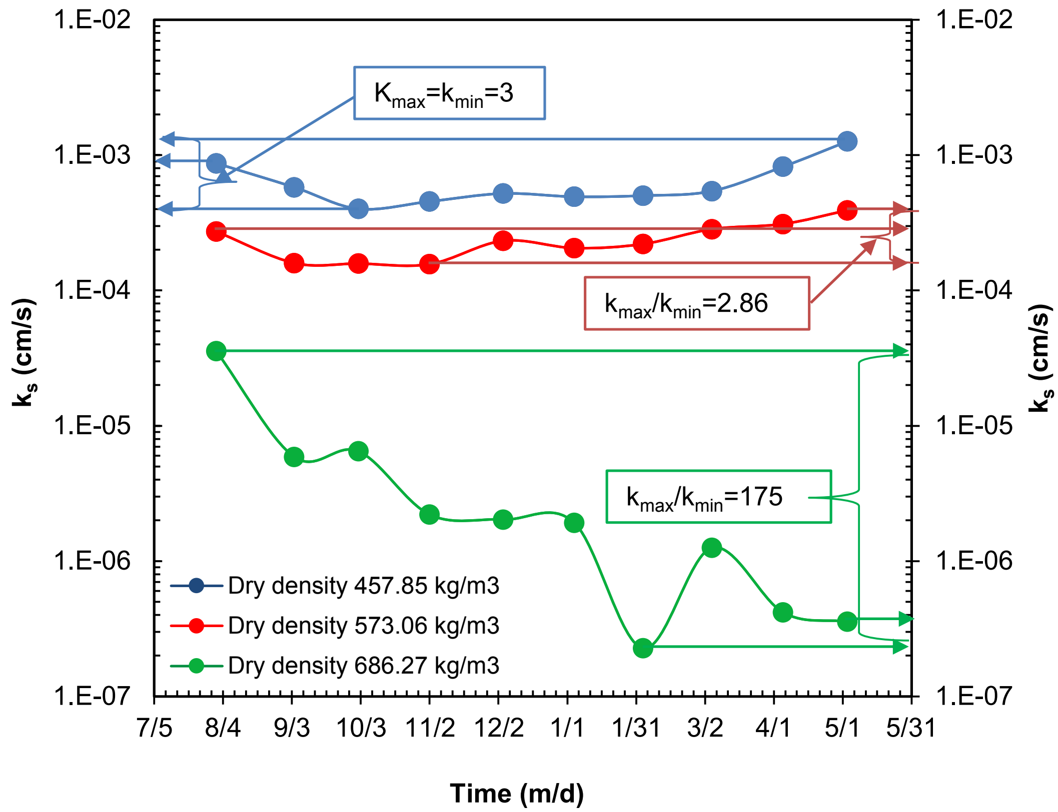

Figure 5-4 Comparison on the fluctuation of permeability for variously compacted waste-B with time

5.2.2 Optimum density from porosity of municipal solid waste

After measuring the permeability of the waste samples, the porosity of all waste samples were measured. Waste have different types of porosity because of its nature. Generally waste can never get saturated in field condition. After exceeding the field capacity of waste, it will start to drain water. Porosity is an important parameter to maintain the flow. In soil, there is direct correlation between porosity and hydraulic conductivity but waste is different material as compared to soil so that it will behave in a different way. But there is some kind of relationship between porosity and hydraulic conductivity. In this study, different type of porosities were also considered to evaluate optimum density.

After saturating, the waste samples were allowed to drain under gravity and left to get stable weight. Porosity utilized by the fluid contents is called as retained/effective porosity. Porosity estimated at the absorptive capacity which is also one of the major important parameter for the bioreactor landfill. Similarly, porosity occupied by the air after drainage.is called as drainable porosity. The tests were carried out in the same permeameters for all samples. Retained porosity and drainable porosity were utilized to evaluate the optimum range of density.

One of the most important parameter to maintain the flow is the hydraulic conductivity of waste and hydraulic conductivity largely depend on porosity of the materials. Generally porous materials have high hydraulic conductivity so that the porosity and hydraulic conductivity parameter related to each other. It is also possible to develop hydraulic conductivity function depending on porosity but many factors need to consider for the waste materials. The porosity of the waste is important parameter not only for liquid movement but also for gas movement. As the waste is degraded it produce gas which need to collected or throw. If the material is highly porous, the gas and liquid movement will be faster. Similarly, leachate generation will also be increased and leachate recirculation will be easier.

If the density of municipal solid waste is too high it will reduce the porosity and can affect the transport liquid and gas from one part to another part. If the moisture cannot circulate along the landfill, waste cannot degrade and remain dry for long period of time. Similarly, leachate will not be produced which is not beneficial for bioreactor landfill operation. Similarly, if the density is low there will be void which can create problem in stability and flow. If there is huge void, there will be result of channel flow and can create ponding inside landfill. There will be pros and cons on both high compaction and low compaction. The current research is focused to evaluate optimum compaction in order to maintain liquid and gas flow inside the landfill.

5.2.2.1 Optimum density from retained porosity

The retained/effective porosity against dry density was plotted for all wastes. The results clearly demonstrated that there was a unique relationship between the retained porosity and dry density. Similar trends were observed for all samples while performing the tests using three different permeameters. Unlike total porosity, retained porosity increased with increasing density and reached a maximum value and started to decrease while increasing density. A tangent intersection method was used to find out the peak point in retained/effective porosity curve. One straight tangent line was drawn from the vertical axis, horizontally parallel to the horizontal axis on the effective porosity curve. The tangent line gave a cutoff point on the curve. A vertical line was drawn from the connecting point of curve and tangent line and extended to horizontal axis to reveal the cutting point. The cutting point where the vertical line hit the horizontal axis was assumed as optimum density. The values of densities obtained from the retained porosity curves for all permeameters were compared. It was surprising to observe almost similar density for all waste through all permeameters results. The procedures of finding optimum density from the retained porosity are shown in Figure 5-5, Figure 5-6 and Figure 5-7.

Different porosities were measured on 25 shredded waste-A, 25 shredded waste-B and 7 unshredded waste samples. Optimum density is evaluated from tangent intersection method for waste-A, waste-B and unshredded waste. The purpose of using many samples was to check and conform the result for optimum density getting for different wastes. The result will be more reliable if the results are verified by many repeated result for different wastes. While using the tangent intersection method, the optimum density were 520 kg/m^3, 490 kg/m^3, 515 kg/m^3 for shredded waste-A through using small, medium and large permeameter device, respectively. The retained/effective porosity was also determined from the horizontal line which crossed the vertical axis. The peak of the curve represented as the maximum retained porosity. The tangent intersection methods determined the maximum retained porosity as 51.2%, 51.2% and 48.8% for waste-A from using small, medium and large permeameter devices, respectively. The determination of optimum compaction and maximum retained/effective porosity for shredded waste-A is shown in Figure 5-5. Similarly, the tangent intersection method determined the optimum density as 490 kg/m^3, 492 kg/m^3, 490 kg/m^3 for shredded waste-B from using small, medium and large permeameter device, respectively. The effective porosity was also determined from the horizontal line which crossed the vertical axis. The tangent intersection methods determined the maximum retained porosity as 51%, 48.7% and 48.8% for shredded waste-B from using small, medium and large permeameter device, respectively. The determination of optimum compaction and maximum retained/effective porosity for shredded waste-B is shown in Figure 5-6. Besides shredded waste-A and waste-B, similar method was used to find out the optimum range of density for unshredded waste-B. The tangent intersection method determined the optimum density as 515 kg/m^3 unshredded waste-B from using large permeameter device. The effective porosity for the unshredded waste-B was also determined from the horizontal line which meet the vertical axis. The maximum retained porosity which was obtained as 46% for unshredded waste-B. The determination of optimum compaction and maximum retained/effective porosity for unshredded waste-B is shown in Figure 5-7.

Basically retained porosity is the porosity filled with liquid at the total absorptive capacity of the MSW. It is always filled with liquid/leachate in the waste. It is easy to understand this term as the ratio of volumetric water retained by waste mass to the total volume of waste mass. It is related to water retaining capacity of the waste mass so that it is an important parameter for the bioreactor landfill. As it is clear that water is needed to enhance the degradation of waste, the retained porosity can play an important role in accelerating the degradation because of retained water. When there is more water, it accelerates the degradation process. Total porosity decreased with increasing density in all types waste whereas the retained/effective porosity was increased initially with increasing density and reached the maximum at some point and again started to decrease with increasing density. This was the unique property of the waste mass which was utilized to evaluate the optimum density. With further increase in density it started to decrease with increasing density. It is also clear that, the range of optimum densities obtained through this process for all waste utilizing all equipment were in the range of 490 kg/m^3 to 520 kg/m^3. As the waste is heterogeneous materials, the difference can be considered as small or almost negligible. The average optimum compaction can be concluded as 505 kg/m^3 while considering retained/effective porosity. The optimum densities and retained/effective porosity for different waste samples are summarized in Table 5-3 and Table 5-4, respectively

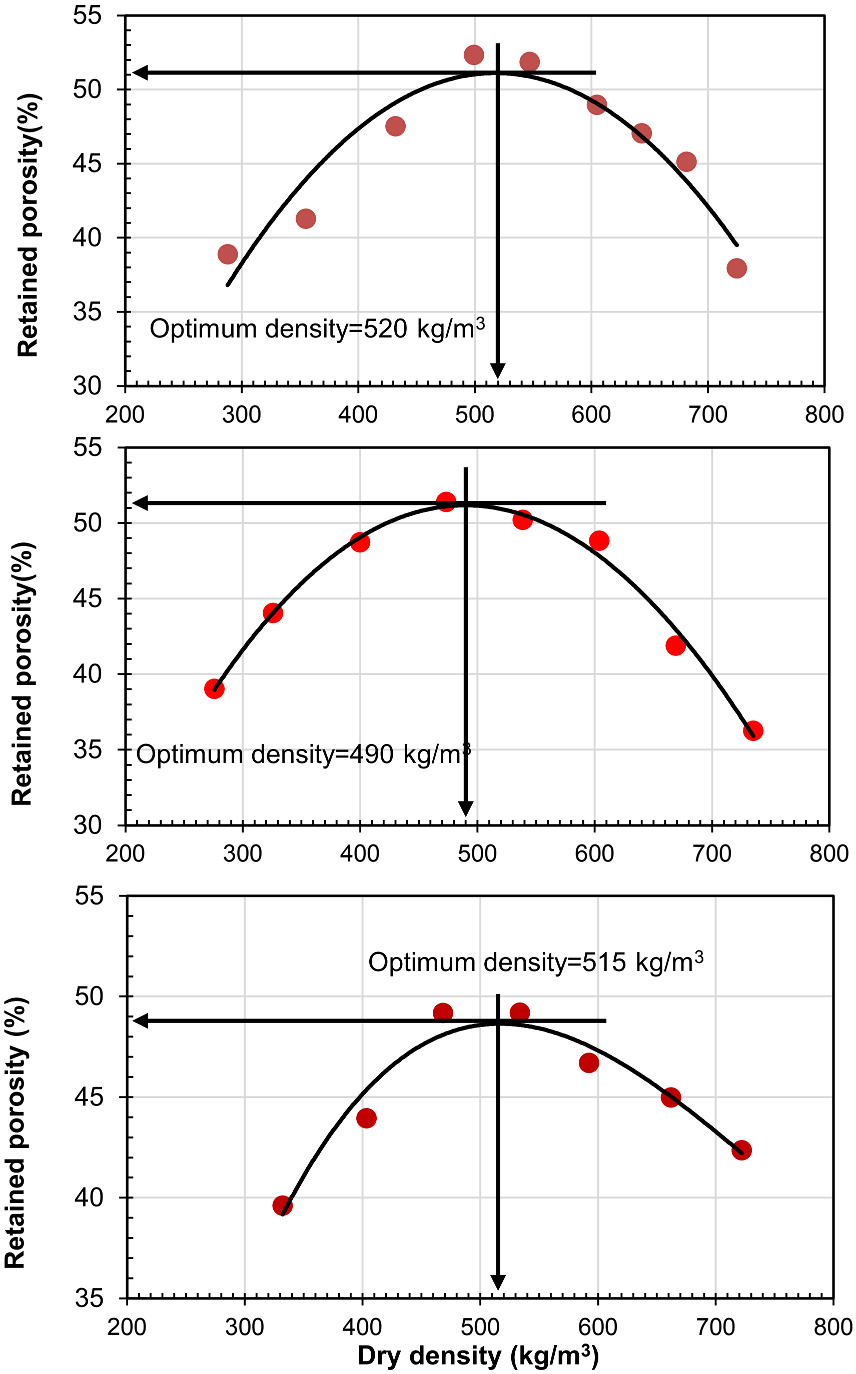

Figure 5-5 Estimation of optimum dry density from retained porosity for shredded waste-A using small, medium and large permeameter

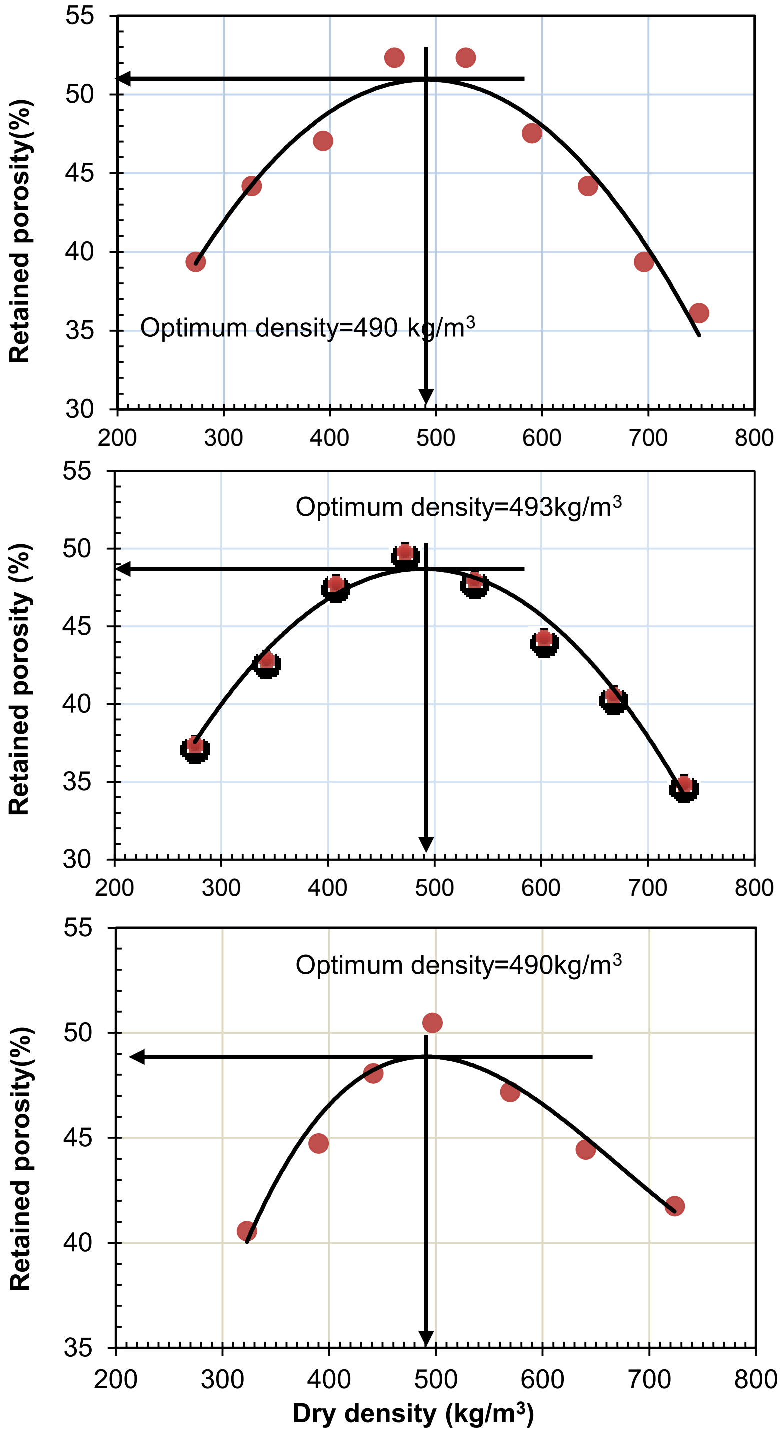

Figure 5-6 Estimation of optimum dry density from retained porosity for shredded waste-A using small, medium and large permeameter

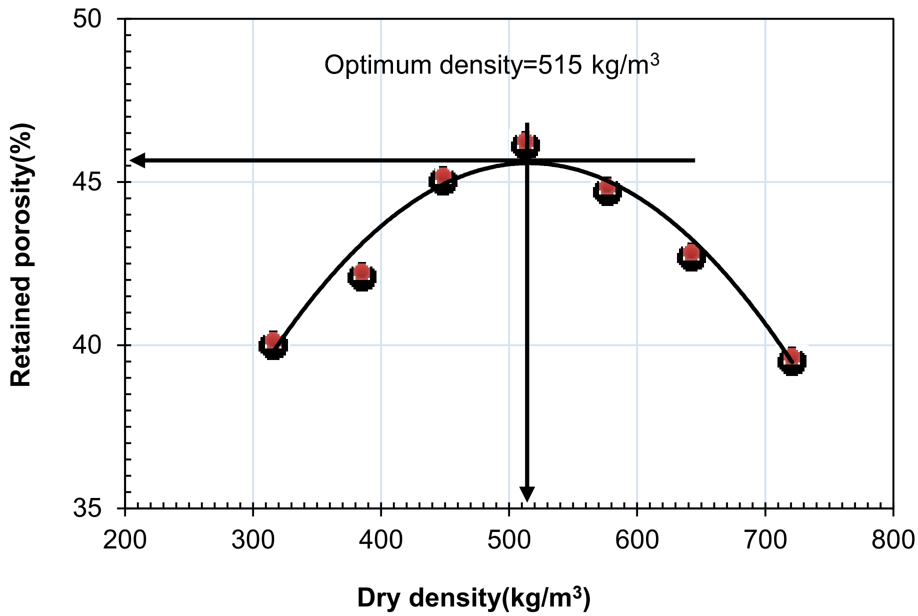

Figure 5-7 Estimation of optimum dry density from retained porosity for shredded waste-A using small, medium and large permeameter

Table 5-3 Summary of optimum densities from retained/effective porosity

Waste type	Optimum density (kg/m^3)		
	From small device	From medium device	From large device
Shredded waste-A	520	490	515
Shredded waste-B	490	492	490
Unshredded waste-B	NA	NA	515

Table 5-4 Summary of retained/effective porosity based on optimum density

Waste type	Retained/effective porosity (%)		
	From small device	From medium device	From large device
Shredded waste-A	51.2	51.2	48.8
Shredded waste-B	51	48.7	48.8
Unshredded waste-B	NA	NA	46

5.2.2.2 Optimum density from drainable porosity

If a fully saturated solid waste is allowed to drain under gravity flow, its water content will decrease as drainable pores empty. It will eventually reach a stable state is known as the field capacity when no further drainage occurs. The amount of freely draining water per unit total volume of waste defines as the drainable porosity. The drainable porosity can also be described as the difference between the saturated volumetric water content and the volumetric field capacity. The drainable porosity was determined directly under laboratory conditions. Knox (1992) estimated drainable porosities of 10-20% on the basis of water balance calculations at several different landfill sites. Beaven (1996) reported the results of a pumping test at a 9 m depth of refuse with a 5-6 m saturated zone, which gave a drainable porosity of 4%. A further pumping test was undertaken on the same refuse after landfilling had increased the depth of waste to 23 m and the thickness of the saturated zone to 6-7 m indicated an increase in drainable porosity to 7%.

The drainable porosity against dry density was plotted for all type of wastes. The results clearly demonstrated that there was a unique relationship between the retained porosity and dry density of MSW which have a similar pattern to the previous studies. Similar trends were obtained between drainable porosity and dry density for all samples while performing the tests using three different permeameters. Drainable porosity decreased with increasing density for all kind of wastes. In lower density range, solid waste had high drainable porosity but it had low value in high density range. It was very clear that the slope was very steep in initial curve and the slope was mild after crossing dry density more than around 500 kg/m^3. It can be observed a turning point on the curve which was assumed as optimum density. Tangent intersection method was used to find out the cutoff point on the curve of drainable porosity against dry density. One vertical straight line was drawn perpendicular from the cutoff point to horizontal axis which was assumed as optimum density. The values of densities obtained from the drainable porosity curves for all permeameters were compared. It was an unique characteristics of waste to observe almost similar values of densities through all permeameters. Similarly, one horizontal straight line was drawn perpendicular from the cutoff point to vertical axis which was termed as minimum drainable porosity required for maintaining flow. If the drainable porosity is low, waste cannot generate leachate because of the high reduction on permeability of solid waste. The procedures of finding optimum density from the retained porosity are shown in Figure 5-8, Figure 5-9 and Figure 5-10.

Drainable porosities were measured on 25 shredded waste-A samples, 25 shredded waste-B samples and 7 unshredded waste samples at various densities level. Optimum density was evaluated from tangent intersection method for all waste in a similar way. The purpose of using many samples was to compare and observe the difference among the results and then finalizing a range of optimum density for maintaining flow. Generally, the result will be more reliable if the results are verified by many repeated result for different wastes. While using the tangent intersection methods, the optimum densities obtained as 520 kg/m3, 490 kg/m3, 515 kg/m3 for shredded waste-A from using small, medium and large permeameter device, respectively. The drainable porosity was also determined from the horizontal line which crossed the vertical axis. The minimum drainable porosities obtained through the methods as 12%, 10% and 12% for waste-A from using small, medium and large permeameter device, respectively. The determination of optimum compaction and minimum drainable porosity for unshredded waste-A is shown in Figure 5-8.

Similarly, the tangent intersection method was applied for waste-B to estimate optimum density and minimum drainable porosity. The optimum densities were obtained as 520 kg/m^3, 490 kg/m^3, 480 kg/m^3 for shredded waste-B through this method while using small, medium and large permeameter device, respectively. The drainable porosity was also determined from joining the horizontal line from the cross section of joints to the vertical drainable porosity axis. The minimum drainable porosities obtained through the methods as 9%, 9% and 9% for waste-B from using small, medium and large permeameter device, respectively. The determination of optimum compaction and minimum drainable porosity for unshredded waste-B is shown in Figure 5-9

Besides shredded waste-A and waste-B, similar method was used to find out the optimum range of density for unshredded waste-B. The tangent intersection method from drainable porosity versus dry density determined the optimum density as 490 kg/m3 unshredded waste-B from using large permeameter device. The drainable porosity for the unshredded waste-B was also determined from the horizontal line extending from the cutoff point of tangents which crossed the vertical axis. The minimum drainable porosity which was obtained as 9% for unshredded waste-B. The determination of optimum compaction and minimum porosity for unshredded waste-B is shown in Figure 5-10.

It is also clear that, the range of optimum density obtained through this process for all waste utilizing all equipment were in the range of 490 kg/m3 to 520 kg/m3. As the waste is heterogeneous materials, the difference can be considered as small. The average optimum compaction can be concluded as 505 kg/m3 while considering retained/effective porosity. Similarly, the minimum drainable porosity is on the range of 9% to 12%. It can be concluded that 10% is the minimum required drainable porosity for the landfill mass. The results clearly demonstrated that the drainable porosity of MSW can be significantly influenced by the increasing density. While considering the liquid movement and distribution, it is a very important parameter connected with capacity of landfill mass to generate leachate. If the drainable porosity is small, the waste cannot generate leachate. If the drainable porosity is high, the leachate will drain quickly so that the recirculation also becomes faster. These results explained there was a correlation between the dry density and drainable porosity of MSW. The general trend was that the drainable porosity decreased with increasing dry density for fresh MSW. Several other researchers also reported the influence of stress on the drainable porosity of MSW. The higher stress in the landfill increases the density. The density will increase with the increasing depth in landfill; therefore, drainable porosity decreases with the increase in the surcharge landfill. The optimum densities and minimum drainable porosity for different waste samples are summarized in Table 5-5 and Table 5-6, respectively

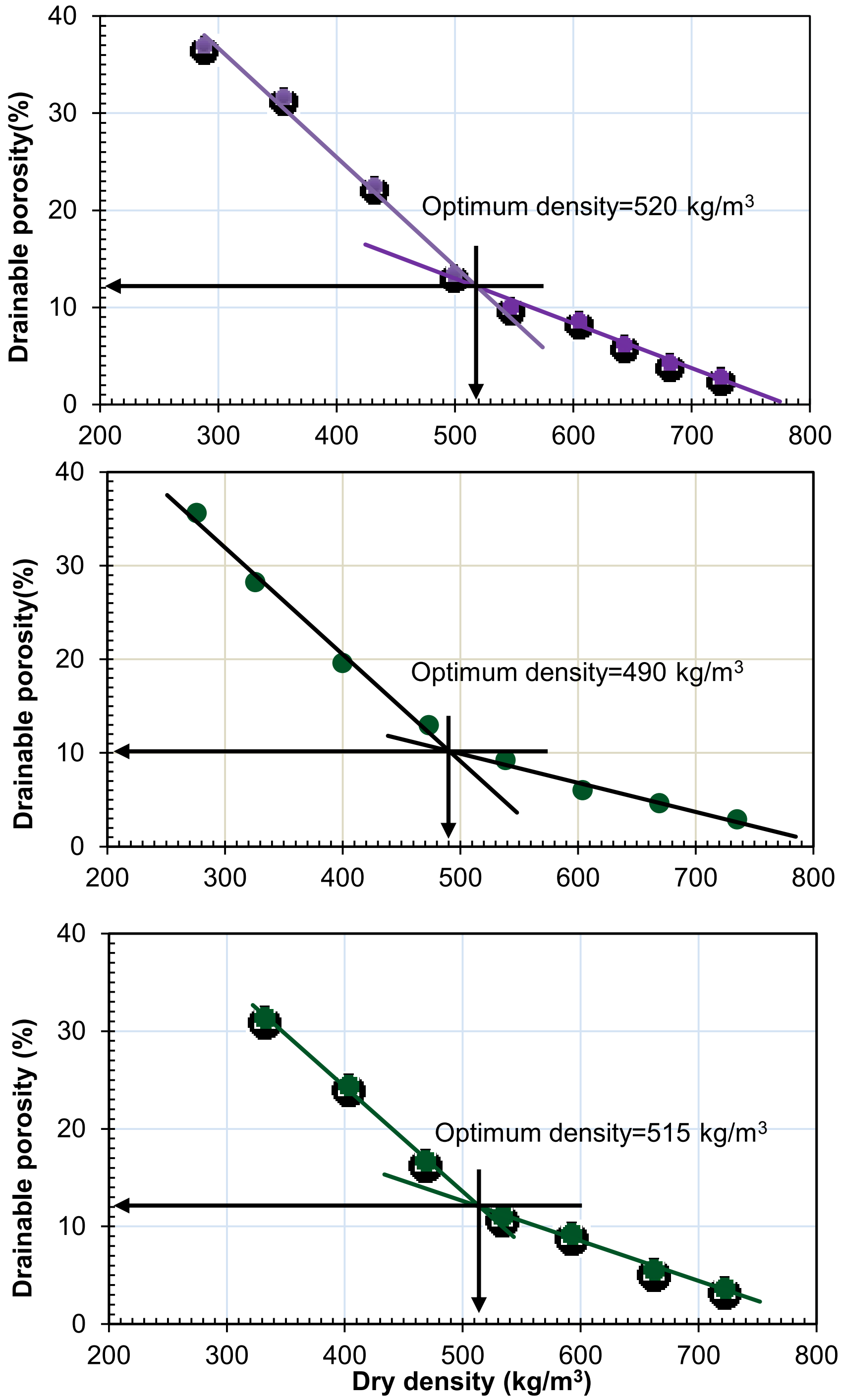

Figure 5-8 Estimation of optimum dry density from drainable porosity for shredded waste-A using small, medium and large permeameter

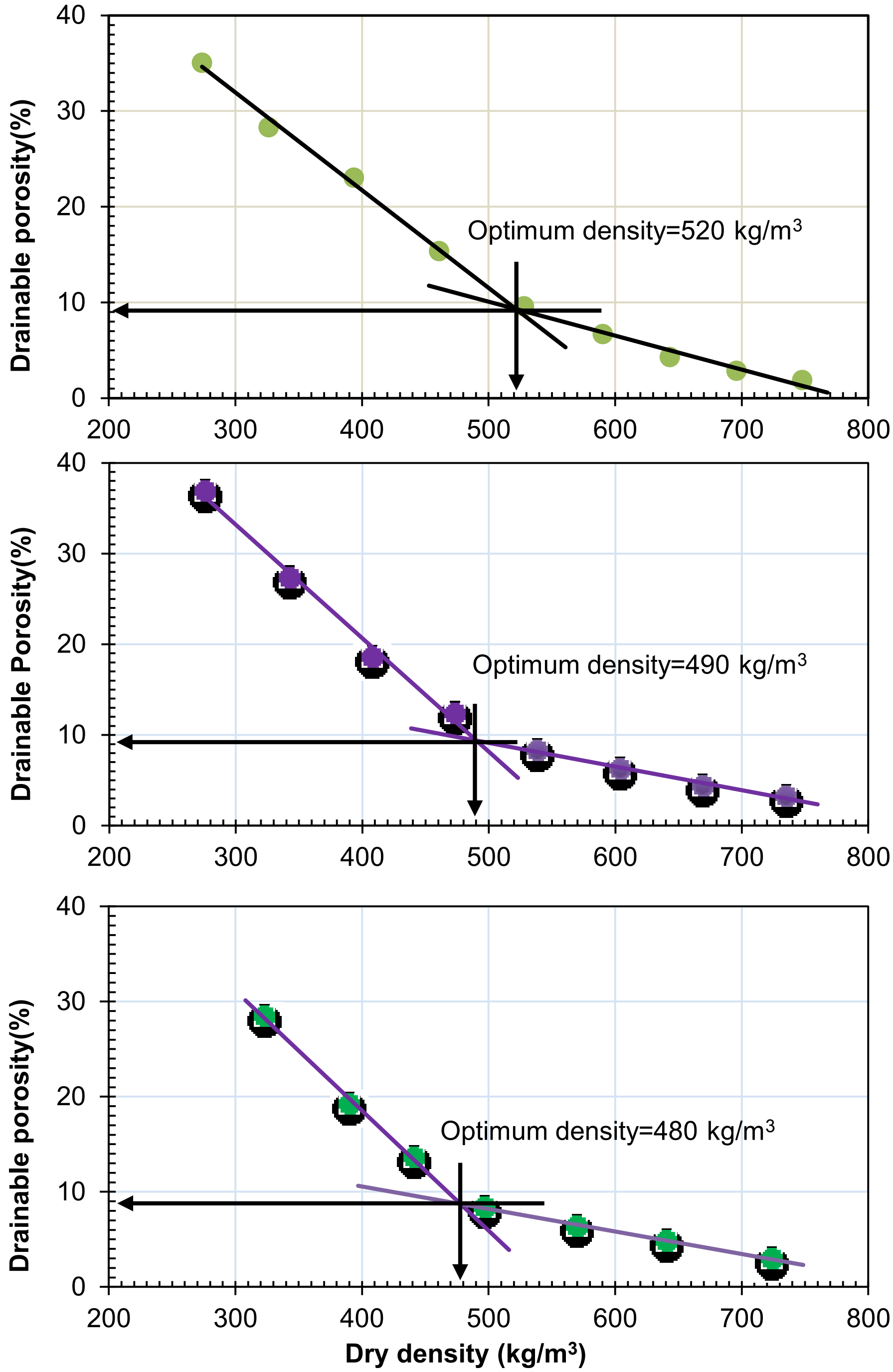

Figure 5-9 Estimation of optimum dry density from drainable porosity for shredded waste-B using small, medium and large permeameter

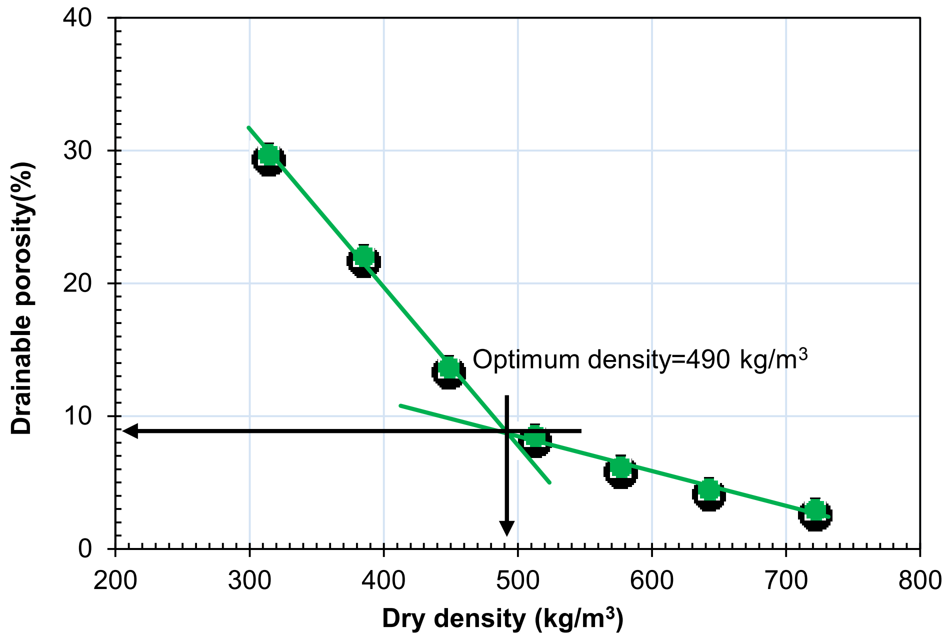

Figure 5-10 Estimation of optimum dry density from drainable porosity for unshredded waste-B using small, medium and large permeameter

Table 5-5 Summary of optimum densities from drainable porosity

Waste type	Optimum density (kg/m^3)		
	From small device	From medium device	From large device
Shredded waste-A	520	490	515
Shredded waste-B	490	492	490
Unshredded waste-B	NA	NA	515

Table 5-6 Summary of minimum drainable porosity based on optimum density

Waste type	Minimum drainable porosity (%s)		
	From small device	From medium device	From large device
Shredded waste-A	12	10	12
Shredded waste-B	9	9	9
Unshredded waste-B	NA	NA	9

5.2.3 Optimum density from degradation of waste

This section provides an overview of the results obtained from one set of experiments conducted to study the effect of compaction on the performance of MSW bioreactors operated with addition of moisture and monitoring gas generation with leachate collection and recirculation to estimate optimum density. Three reactors were compacted with exactly similar waste by the composition and monitored for around 1 year.

The concept of a bioreactor is the addition of moisture that stimulates microbial activity by providing better contact between the waste and microorganism via solvent medium. Leachate movement through the waste mass is controlled by hydraulic conductivity of the waste which strongly depends on anisotropy heterogeneity, partial saturation and level of compaction of solid waste. Currently landfill operators tend to compact the waste as much as possible to accommodate greater amounts of waste within the same space. However, higher density decreased hydraulic conductivity, which eventually inhibit downward moisture movement and thus ultimately inhibit the degradation of the solid waste mass. Therefore, identifying the optimum compaction range for bioreactor landfill operation is considerably important to enhance the degradation. If the density of municipal solid waste is too high the liquid cannot move around the waste mass and may remain dry for many years. Similarly if the density of municipal solid waste is too low, there will be ponding inside the landfill and may slowdown the degradation process. Considering these situation, it is necessary to find out the range of optimum density which can accelerate degradation process without affecting the required flow of liquid and gas. Previous studies has a limitation on finding the effect of compaction on the degradation but mostly focused on the effect on hydraulic conductivity. It is very important to understand the effect of compaction not only on the liquid movement but also on the gas generation and leachate production. The proper density was estimated based on the gas generation and leachate production for the fresh solid waste in the following sections.

5.2.3.1 Optimum density based on gas generation

There was not available any study how the compaction might effect on the degradation and gas generation from municipal solid waste. But there was tendency to over compact the waste in landfill by owner. In order to cope with this issue, the current study has focused on the compaction effect on the degradation of solid waste. Three laboratory scale compacted bioreactor cells were designed to study about the compaction effect on the degradation level. Biodegradation of the municipal solid waste (MSW) were carried out to investigate the dependency of degradationon the compaction level. Three compacted MSW cells named as reactor R1, R2 and R3 were purposely developed to investigate the biodegradation of MSW because of the compaction effort. Solid waste having exactly similar kinds of composition were compacted in the same volume of laboratory bioreactor cell by maintaining the dry densities of 457.8 kg/m^3, 572.1 kg/m^3 and 686.3 kg/m^3 for reactor R1, R2 and R3, respectively. The experimental results indicated that the biodegradation of the MSW is highly dependent upon the compaction level. The reactor R1 which has lowest density 457.8 kg/m^3 produced maximum methane gas while reactor R3 which had highest density 686.3 kg/m3 produced low methane gas.

The total methane gas generation for all compacted samples is given in Figure 5-11. The reactor R1, R2 and R3 produced total methane gas as 338 L, 319 L and 122 L, respectively within 1 year from same volume of reactors. There was almost similar amount of methane production from two low and medium compacted waste but the production of methane was negligible as compared to other two compacted waste samples. The generation of methane can also be expressed in terms of volume of equipment. The volume of each reactors were exactly same and waste samples were compacted at different densities. The same volume of reactor contain same volume of waste but different mass. While calculating the production of methane on volumetric basis, the reactors R1, R2 and R3 produced 23.8 L/L, 22.5 L/L and 8.6 L/L, respectively which is shown in Figure 5-11.

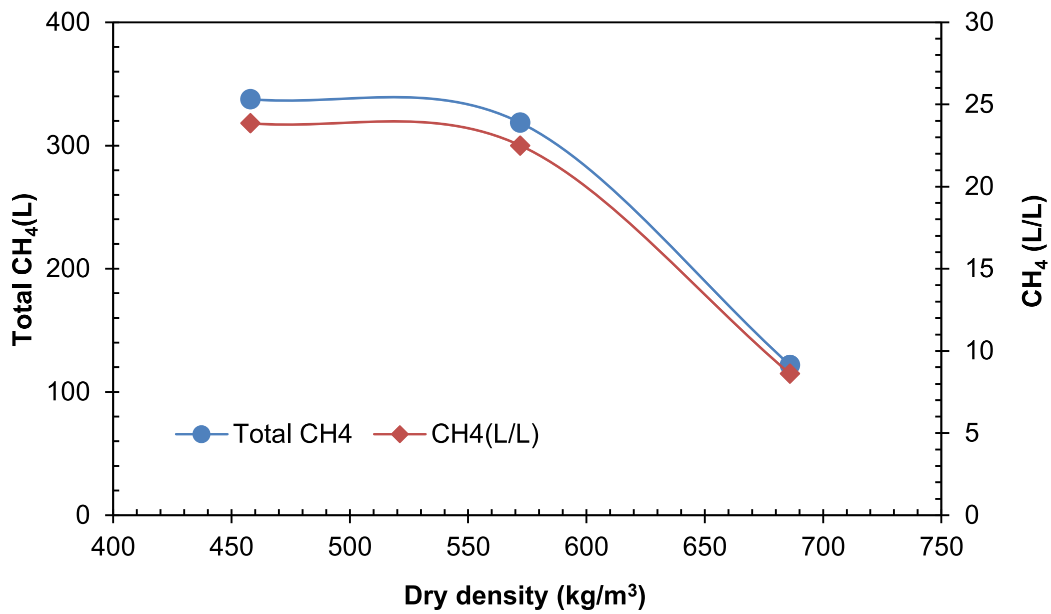

Figure 5-11 Comparison of methane gas generation among reactors R1, R2 and R3

The rate of methane gas generation for all compacted samples is given in Figure 5-12. The reactor R1, R2 and R3 has reached peak of methane gas generation 315 mL/kg/day, 245 mL/kg/day and 37 mL/kg/day, respectively. It is clear that high compaction has slow down the gas generation rate. Basically the reactor R3 has no any distinct peak value. It has almost constant rate which is very low as compared to other reactors gas generation rate. All reactors were operated in identical condition except leachate recirculation. Leachate also tried to recirculate in constant rate to all reactors but compaction reduced the leachate absorbance to the waste.

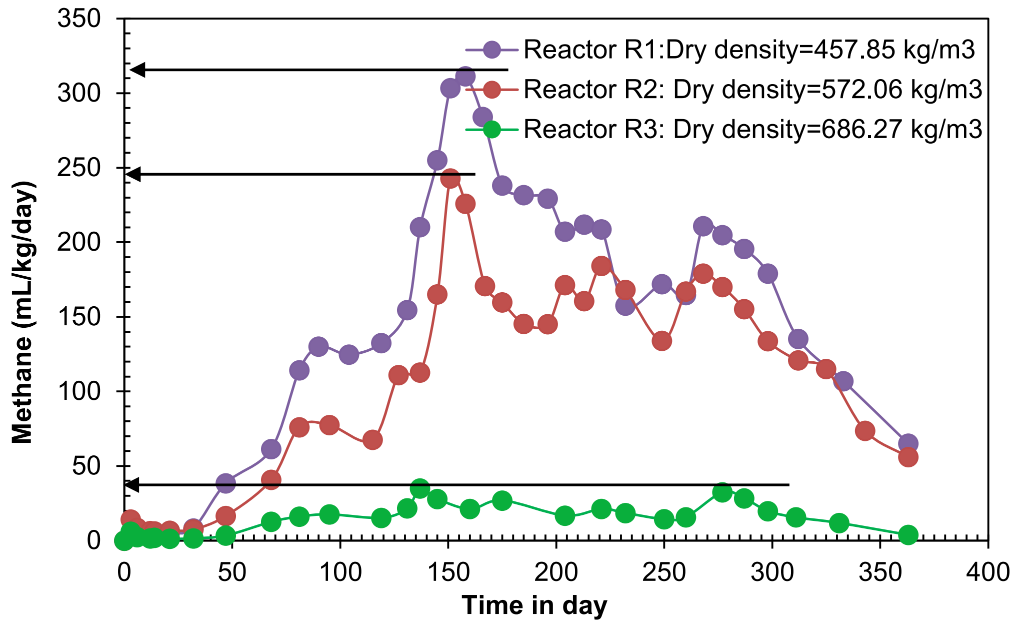

Figure 5-12 Comparison of methane gas generation rate on reactors R1, R2 and R3 at different densities

The cumulative methane gas production for all reactors R1, R2 and R3 is given in Figure 5-13. The overall volume of methane gas produced was much higher in the reactor R1 than in the reactor R2 and R3. In addition, overall volume of methane gas was very low in the reactor R3. There was a continuous production of methane gas in reactors R1 and R2 which showed almost similar production. The cumulative rates of methane gas from reactors R1, R2 and R3 were 52L/kg, 39 L/kg and 12.5 L/kg, respectively one year period of time. It is clear that compaction had a great impact on the gas generation which is shown by the Figure 5-13 and Figure 5-14 . The compaction ratio of reactors R1: R2 and R3 were in the ratio of 1:1.25:1.5, respectively. Similarly, it was observed that the gas generation ratio for reactors R1, R2 and R3 are 4.16:3.12:1, respectively.

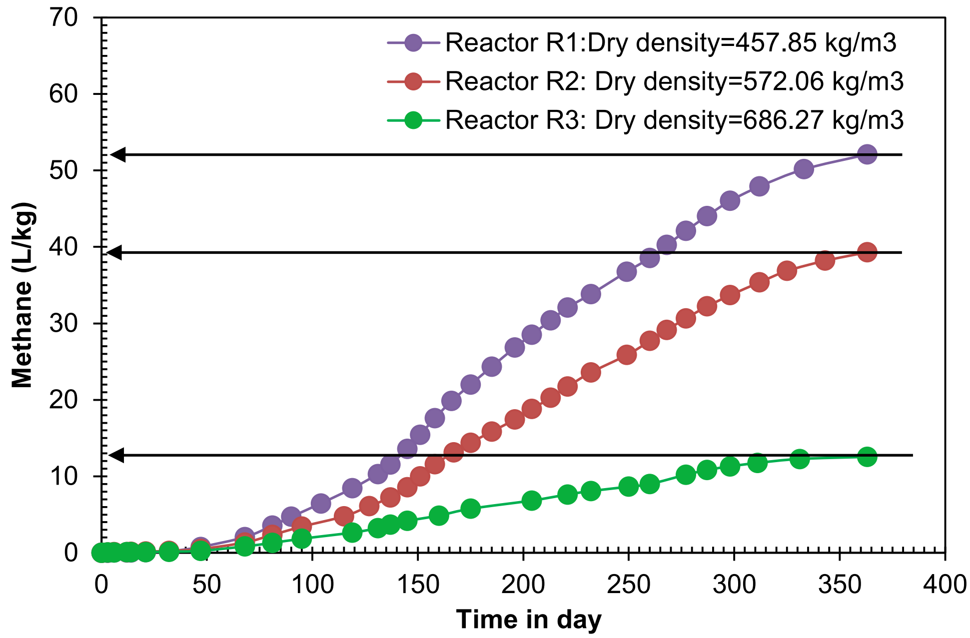

Figure 5-13 Comparison of cumulative methane gas generation among different compacted waste density

It is very clear that compaction has a huge impact on the degradation of municipal solid waste which is not beneficial for the bioreactor landfill operation. Since the purpose of bioreactor is to enhance the degradation and reducing the post closure time. While considering the gas generation low compaction is good for degradation. But the current study has the limitation because the comparisons was within three densities. It is insufficient to conclude the optimum density based on three reactors data but it can be concluded that high compaction after 572.1 kg/m^3 is not favorable for degradation. Even though waste degrades in high compaction but it cannot produce methane gas. There was a high variation on the methane gas generation rate from reactors which were mostly operated in low density. Most of the previous reactors contained hand compacted waste which should have low density. In order to understand the detail about the compaction effect on the gas generation and leachate production, several reactors will be needed. If there are many points, it might give a parabolic curve for the gas generation versus density. It can be concluded from this study, the compaction might be favorable when the densities are on the range of 457.8 kg/m^3 to 572.1 kg/m^3.

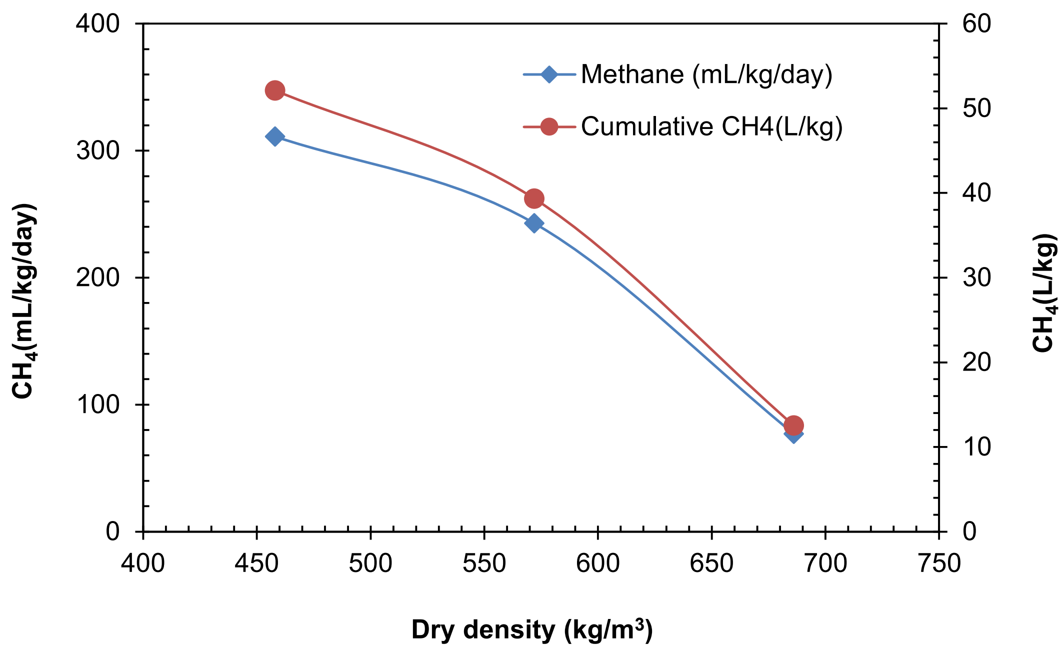

Figure 5-14 Comparison of methane gas generation rates at differently compacted municipal solid waste

Another most important is the stability issue of the landfill slope. The current study has not been focused on the strengths of solid waste at different compaction level. Although low compacted waste produce maximum methane gas, it might not be benefit to compact the waste with very low density. If the density is very low, there will be stability problems and landfill slope might fail. Several landfill slopes failed due to having low shear strength. When water is added it will reduce the shear strength of waste. Besides moisture content, degradation also reduces the strength. There are so many factors which initiates the failure of slope if the density is low. It is not very clear what might be the gas generation in lower densities than the current density of reactor R1, R2 and R3. It is necessary to come up for optimum density by considering gas generation with shear strength for stability. This current study conclude the optimum density might be in the range of 458 kg/m^3 to 572 kg/m^3.

5.2.3.2 Optimum density based on leachate

The role of leachate recirculation had proven to directly impact the gas generation potential. Increased gas production was directly related to leachate recirculation process and frequency. The leachate circulation and generation capacity highly depend on compaction level. If the density is low, leachate circulation will be easy and quicker due to high permeability of solid waste. The low density also might have pros and cons for degradation and stabilization. The low gas production volumes might occur due to the washout of the substrate and essential nutrients, thus limiting the microbial activity and waste stabilization. But in the meantime it will increase the leachate generation capacity. Similarly too much compaction will reduce the efficiency of circulation and generation of leachate. The compaction will lead to higher density which will reduce permeability and thus leachate generation.

The experimental study had been focused on the efficiency of leachate circulation and generation due to compaction level. The compaction highly reduced the absorptive capacity of the solid waste and thus leachate generation. The leachate recirculation rate was tried to maintain the same to all reactors depending on the mass. Certain amount of leachate was tried to circulate each time to all reactors but the leachate absorbent rate was different. The reactor R1 absorbed huge amount of water and leachate easily each time. It took very short time to circulate the leachate on the low compacted waste on reactor R1. The reactor R2 absorbed the leachate/water in the same ratio as reactor R1 in the few initial month but the absorbing capacity was decreased after 2 month. When the leachate was tried to recirculate in the same ratio, reactor R2 did not absorb all leachate after two month. Reactor R3 did not take all water/leachate when it was tried to circulate in the same ratio as R1 and R2. The water/leachate absorbent rate was very low a lot just after 1 month. It took very less amount of water and leachate so that the gas generation was also very low. Similarly the leachate generation rate was also very low which was almost negligible as compared to reactor R1 and reactor R2. It is also one of the most important criteria to measure the efficiency of bioreactor landfill operation. It is assumed that if the leachate generation is high, gas generation will also be high. While considering this criteria to estimate the optimum density level, the density should be even less than that of reactor R2 i.e. dry density of 572.1 kg/m^3. It can be concluded that the optimum density should be lower than 572.1 kg/m^3 and which could be in between the density of reactor R1 and reactor R2.

Chapter 6

Conclusion and recommendations

6.1 Introduction

The fundamental aspect in the operation of a bioreactor landfill is the addition of water and/or the recirculation of leachate into the landfill's waste mass. Understanding the moisture distribution within a bioreactor landfill is essential for the design and operation of the leachate recirculation system. However, the distribution and movement of liquid is greatly affected by the density of the waste mass. The fundamental principles of operating the landfill as the bioreactor are to enhance decomposition process and to ensure landfill stability and safety as compared to conventional landfills. Since recirculation of liquid into waste is important in a bioreactor landfill, the understanding of the hydraulic conductivity and porosity become necessary. Basically, hydraulic conductivity of any materials is dependent on its porosity. The waste is extremely heterogeneous materials and there exist both saturated and unsaturated zones that make it more complex to understand and quantify the liquid flow in the waste. A further complication arises due to the change in porosity as the degradation process occurs. There have been a numerous studies on the hydraulic conductivity of the MSW and the reported a big range of hydraulic conductivities from 10^{-2} to 10^{-7} cm/sec. The large variation in the hydraulic conductivity of the waste may be due to the variation in composition, density, water content, testing method, size of equipment and several other factors. Although several studies were conducted in the past to find the hydraulic conductivity, there was lack of systematic work to find out the causes of fluctuation of hydraulic conductivity. Also most of the past studies reported change in hydraulic conductivity with density but there is still lack of finding the optimum density in order to increase the efficiency of flow and enhance the degradation. In the recent years, there has no any specific guidelines for the compaction level in order to increase the methane gas and leachate generation and thus to increase the efficiency of bioreactor operation. The main objective of this research was to evaluate the optimum range of density for the bioreactor landfill operation.

The whole idea of this research was to find out the effect of compaction on the hydraulic parameters, effect of compositional variation on the hydraulic parameters and dry density. The research has also been focused on the properties of individual components of the MSW in order to better understand the waste. Since the MSW is the combination of different waste components, its properties are controlled by the component's content presence in MSW. A detail experimental tests were performed in order to understand the hydraulic properties of MSW. Similarly, the research has focused on the effect of compaction on the degradation pattern of MSW. Further, an attempt was made to estimate an optimum compaction for the bioreactor landfill operation by considering hydraulic parameters and degradation.

6.2 Summary and conclusions

The following results and conclusions are based on the findings from this study:

1. Based on the experimental results, shredded waste-A had 32.9% paper, 27.9% plastic, soil and fines content 12.0% yard/wood 10.2%, C&D 7.6%, textiles 4.6%, Styrofoam and sponge 2.4%, metal 2 %, glass 0.3%, food 0.1%. Similarly waste–B included 41.4% paper, fine particles and soil 21.1%, plastic 17.7%, yard and wood 6.4%, food 3.6%, metal 3.4%, Styrofoam and sponge 3.1%, textiles 2.1%, C&D 0.72%, glass 0.48%. The waste-A had maximum dry density of 376 kg/m^3 at 62% optimum moisture content. Similarly shredded waste-B resulted in a maximum dry density of 410 kg/m^3 at 58% optimum moisture content.

2. Hydraulic conductivity of MSW was greatly influenced with the density which is basically due to lowering in porosity. Hydraulic conductivity for various waste samples decreased while increased dry density. MSW was greatly influenced with the density Different type of waste showed the similar nature. The current value of permeability are within the range of previous studies.
3. The permeability are greatly affected by the gas formation. The high drop in permeability was observed when no gas was released. Similarly permeability increased significantly while gas was released with flushing. The reduction in permeability might be due to conversion into unsaturation state in waste.
4. Total porosity of MSW was greatly influenced by the increase in density. Total porosity was decreased continuously while increasing density. When the density is increased the particle of waste come close to each other and reduce the space available among the particles. Different type of waste showed the similar nature.
5. Retained porosity is the porosity occupied with retained water inside solid waste. It is the volumetric water content occupied by waste. Retained/effective porosity of MSW was greatly influenced by the increase in density. Initially the retained porosity increased, reached a peak value and again started to decrease while increasing the dry density. Different type of waste showed the similar nature. The maximum value of the retained porosities were around 46% to 52% while using different device for various waste samples.
6. Drainable porosity is the porosity occupied with air inside solid waste after draining water. Drainable porosity of MSW was also greatly affected by the increase in density. Drainable porosity decreased significantly while increasing density. Drainable porosity is related to the liquid draining capacity of waste so that it influenced the leachate generation from waste.

7. The correlation between total porosity and dry density can be expressed by the relation as, $y = -6E^{-05} \times x^2 - 0.0084 \times x + 80.822$ where y is the porosity in percentage and x is the dry density in kg/m^3.
8. The correlation between retained porosity and dry density can be expressed by the relation as, $y = -0.0002 \times x^2 + 0.2205 \times x - 6.5551,$ where y is the porosity in percentage and x is the dry density in kg/m^3.
9. The correlation between drainable porosity and dry density can be expressed by the relation as, $y = -4\,E^{-8} \times x^3 + 0.0002 \times x^2 - 0.2611 \times x + 92.303$ where y is the porosity in percentage and x is the dry density in kg/m^3.
10. The moisture retention capacity and full saturation decreased with increasing in dry density of fresh MSW. Moisture content is directly related to porosity. The porous materials have high saturated moisture content whereas absorptive capacity is related to liquid retained capacity of waste. If the waste contains water nonabsorbent materials, it will have low absorptive capacity. Several previous studies also reported the influence of density on the moisture content of MSW.
11. The moisture contents at full saturation decreased with density which can be expressed by the relation, $y = 0.0008 \times x^2 - 1.212 \times x + 532.9$, where y is the saturated moisture content in percentage and x is the density in kg/m^3
12. The water retained capacity decreased with dry density which can be expressed by the relation $y = -0.1894 \times x + 190.77$ where y is the water retention capacity in percentage and x is the dry density in kg/m^3.
13. Permeability in municipal solid waste is mainly dependent on the density, pore size and its geometry. Besides these the permeability is also highly affected by composition, shredding of waste. The moisture content at compaction, length to height ratio of sample, hydraulic head also can affect in some way.

14. There are several factors which can influence the degradation of municipal solid waste. These parameters include stress/density, moisture content, nutrient content, pH level, bacterial content, oxygen concentration, and temperature, composition and flushing system. Density is one of the most important parameter which effect on gas generation and degradation has not been considered yet.
15. The rate of methane gas generations for compacted samples for reactor R1, R2 and R3 has reached peak of methane gas generation 315 mL/kg/day, 245 mL/kg/day and 37 mL/kg/day, respectively. The compaction has adverse effect on the gas generation from MSW. There is not huge difference between gas generation from reactors R1 and R2.
16. The rate of methane gas generations for compacted samples for reactor R1, R2 and R3 produced methane gas 23.85 L/L, 22.5L/L and 8.61L/L, respectively from similar volume of reactors.
17. The reactor R1, R2 and R3 produced 338 L, 319 L and 122 L volume of methane gas in one year period of time. The cumulative rates of methane gas from reactors R1, R2 and R3 were 52L/kg, 39 L/kg and 12.5 L/kg, respectively.
18. The compaction ratio of reactors R1: R2 and R3 are in the ratio of 1:1.25:1.5, and it was observed that the gas generation ratio for reactors R1, R2 and R3 are 4.16:3.12:1, respectively.
19. The time of pH changing was less in the lowest compacted reactor and the time was increased with the compaction level. The pH changing time from acidic to basic was around 110 days for the lowest compacted sample, 125 days for the medium compacted samples and 150 days for the highly compacted samples
20. The total circulation of water/leachate was 28L in reactor R1, 20.5L in reactor R2 and 6.5 L in reactor R3, respectively even though leachate was tried to recirculate in the

same ratio of weight. The compaction also affected recirculation capacity on the waste sample. The time of recirculation was less for low compacted and it increased with density. It seems that too much compaction might not be helpful for the circulation.

21. The total leachate generations through reactors R1, R2 and R3 were 25 L, 15 L and 3 L in reactor R3, respectively. When the recirculation was affected by compaction, it affected on the leachate generation.
22. While performing the permeability tests on the reactors, the entrapped air was always removed by applying water from the bottom of the samples. This process was followed all the times in order to fully saturate the samples by removing air inside waste. This process removes almost all generated gas that was stored in void space inside the samples. The coefficients of permeability varies from 8.72×10^{-4} cm/sec to 4.01×10^{-4} cm/sec, 2.73×10^{-4} cm/sec to 1.56×10^{-4} cm/sec and 3.78×10^{-5} cm/sec to 2.27×10^{-7} cm/sec for R1, R2 and R3 reactors. The degradation level in reactors R1 and R2 were quite high as compared to reactor R3. Even though there was high degradation of solid waste in reactors R1 and R2 but the hydraulic conductivities were not varying but slightly increased after 8 months. There was not much degradation on the reactor R3 but the hydraulic conductivity decreased every month which indicated that hydraulic conductivity might decrease due to accumulation of gas rather than degradation by itself.
23. Generally, the color of waste was very clear and had clean in fresh state. As the degradation continued the color changed into black which was clearly observed in reactors. The degraded waste seemed blacker as compared to less degraded waste. When the reactors were dismantled, the reactor R3 which showed less degradation and the color of waste in R3 was very clear and clean in deeper part of reactor. The

reactors R1 and R2 reactors had black color waste and seemed highly degraded. The color of waste in the same reactors were also varied with the depth. The top part of the waste was blacker as compared to the bottom and this was more distinctly observed in reactor R3.

24. The change in volatile solids in R1, R2 and R3 reactors as compared to fresh waste were found to be 9.96%, 9.00% and 6.35%even after operating the reactors for a period of 12 months. The reduction in volatile solids were 13.01%, 11.76%, 8.30% for the reactors R1, R2 and R3, respectively. The reduction in volatile solids was very low in the current study as compared to previous studies
25. The three different devices resulted almost similar optimum density and the corresponding hydraulic conductivity for waste-A. The optimum density was 512 kg/m^3 and the corresponding value of hydraulic conductivity based on optimum density was 1.2E-03 cm/s through the small permeameter. The optimum density was 520 kg/m^3 and the corresponding value of the hydraulic conductivity based on optimum density is 9.0E-04 cm/s through the medium permeameter. Similarly the optimum density was 510 kg/m^3 and the corresponding value of the hydraulic conductivity was 1.10E-03 cm/s through the large permeameter.
26. Similarly the similar three devices resulted almost similar optimum density and the corresponding hydraulic conductivity for waste-B. The optimum density was 482 kg/m^3 and the corresponding value of hydraulic conductivity based on optimum density was 1.1E-03 cm/s through the small permeameter. The optimum density was 511 kg/m^3 and the corresponding value of the hydraulic conductivity based on optimum density is 7.0E-04 cm/s through the medium permeameter. Similarly the optimum density was 500 kg/m^3 and the corresponding value of the hydraulic conductivity was 1.0E-03 cm/s through the large permeameter.

27. In the same the optimum density was 490 kg/m^3 and the corresponding value of hydraulic conductivity was 9.0E-04 cm/s for unshredded waste-B through the large permeameter.
28. The permeability reduction ratio on two compacted samples R1 and R2 were 3 and 2.86, respectively. These two samples were more degraded samples which was validated by the huge gas generation from these samples. The permeability reduced in small ratio in these two samples although high degradation occurred. Whereas, the permeability reduction ratio on most compacted reactor R3 was 175 although less degradation occurred in this sample. This concluded that, if the degradation within these range, flow will not be that much affected due to degradation on time. It can be concluded, the density should be less than 573.1 kg/m^3 while considering the long term flow for the bioreactor landfill.
29. While using the tangent intersection method on retained porosity versus dry density, the optimum density are 520 kg/m^3, 490 kg/m^3 and 515 kg/m^3 for shredded waste-A from using small, medium and large permeameter device, respectively. The effective porosity is also determined from the horizontal line which meet the vertical axis. The peak of the curve represents the maximum retained porosity which were as 51.2%, 51.2% and 48.8% for waste-A from using small, medium and large permeameter device, respectively.
30. Similarly the same tangent intersection method determined the optimum density as 490 kg/m^3, 492 kg/m^3, 490 kg/m^3 for shredded waste-B from using small, medium and large permeameter device, respectively. The effective porosity is also determined from the peak of the curve which were 51%, 48.7% and 48.8% for shredded waste-B from using small, medium and large permeameter device, respectively.

31. Besides those, the optimum density was 515 kg/m^3 and the corresponding peak retained porosity was 46% for unshredded waste-B while using large permeameter device.
32. While utilizing the drainable porosity, the optimum density obtained as 520 kg/m^3, 490 kg/m^3 and 515 kg/m^3 for shredded waste-A from using small, medium and large permeameter device, respectively. The minimum drainable porosities obtained through the intersection methods were 12%, 10% and 12% from using small, medium and large permeameter device, respectively.
33. Similarly, the application of tangent intersection methods for drainable porosity versus dry density gave the optimum density as 520 kg/m^3, 490 kg/m^3, 480 kg/m^3 for shredded waste-B from using small, medium and large permeameter device, respectively. The minimum drainable porosities obtained through the intersection methods as 9%, 9% and 9% for waste-B from using small, medium and large permeameter device, respectively.
34. Beside shredded waste, the tangent intersection method on drainable porosity versus dry density determined the optimum density as 490 kg/m^3 for unshredded waste-B. The minimum drainable porosity 9% was obtained through intersection for unshredded waste-B.
35. It is insufficient to conclude the optimum density based on the three different densities but it can be concluded that high compaction after 572.1 kg/m^3 was not favorable for degradation because of the high reduction on gas generation and leachate production. Even though waste degrades in high compaction but it cannot produce methane gas.
36. Similarly the leachate generation rate was also very low from reactor R3 which was almost negligible as compared to reactors R1 and reactor R2. It also one of the most

important criteria to measure the efficiency of bioreactor landfill operation. It was general assumption that if the leachate generation is high, gas generation will also be high. While considering this criteria to estimate the optimum density level, the density should be even less than that of reactor R2 i.e. the dry density of 572.1 kg/m^3.

6.3 Recommendations for future studies

The following recommendations are suggested for future studies:

1. Further research needs to be performed to validate the optimum density for different waste at different composition in predicting methane generation rates as compared to the current study. A dataset with varying climatic conditions and waste compositions should be considered. There should be additional research to identify the correlation between laboratories to field scales gas generation for different compaction ranges.
2. Further research needs to be performed for variable waste from very low to very high density range to come up with the exact effect of density on the gas generation and degradation of waste. Multiple lab scale reactors should be used to verify the exact effect of compaction.
3. Different waste compositions can be considered with multiple reactors could overcome the limiting conditions of the current study.
4. It was observed in this research that the methane generation was based on just the one composition of waste. Therefore several research should be performed in order to develop a model for density effect on the gas generation, leachate generation and degradation of waste. This will improve the applicability of this research towards the field condition.
5. Further research is necessary to understand the impact of compaction on the variation of hydraulic conductivity and porosity over long time. It should be done on

the various density ranges from low (200 kg/m^3) to high (800 kg/m^3) in order to understand the better picture which will be applicable toward the filed condition.

6. In this study, the effect of temperature was not considered on the gas generation and degradation. Further research is necessary to correlate the actual temperature effect to validate the current study.
7. Basically, the optimum density should be determined considering three factors as gas generation from waste, flow efficiency and variation from fresh to degraded waste and stability of landfill waste slope and stabilization of waste.
8. The current study was heavily focused on the flow efficiency and variation over high range of densities. Similar kind of study should be performed for the gas generation and degradation of waste over high range of densities.
9. Besides degradation and flow, stability of landfill waste is one of the most important concern. Bioreactor landfill might be failed if proper caution is not taken place while designing. Several multiple research should be performed to investigation the relationship among compaction, moisture variation and strength parameter. Since leachate is recirculated periodically on landfill, it is utmost important to find out the effect of moisture content on the shear strength parameter. For the successful operation of bioreactor landfill, there should be harm also around the surrounding area. To ensure this, stability factor should be also considered while compacting the fresh waste on landfill.
10. Research should be performed to find out the most critical component for the shear strength parameter and effect of compaction and degradation on shear strength parameters. This will help in order to improve the stability of waste slope which is the most important for bioreactor landfill operation.

References

Ağdağ, O. N., & Sponza, D. T. (2005). “Anaerobic/aerobic treatment of municipal landfill leachate in sequential two-stage up-flow anaerobic sludge blanket reactor UASB)/completely stirred tank reactor (CSTR) systems”. Process Biochemistry, 40(2), 895-902.

Al-Kaabi, S. (2007).”Effect of salinity on biodegradation of MSW in bioreactor landfills” (Doctoral dissertation, University of Ottawa).

Al-kaabi, S., Van Geel, P. J., & Warith, M. A. (2009). “Effect of saline water and sludge addition on biodegradation of municipal solid waste in bioreactor landfills”. Waste Management & Research, 27(1), 59-69.

Angelidaki, I., Alves, M., Bolzonella, D., Borzacconi, L., Campos, J., Guwy, A., Kalyuzhnyi, S., Jenicek, P., and Lier, J. B. (2009). "Defining the biomethane potential (BMP) of solid organic wastes and energy crops: a proposed protocol for batch assays." Water Science and Technology, 59, 927-934.

ASTM, E. 0. (2006). 06a Standard Test Method for Measurement of Fracture Toughness. Annual Book of Astm Standards. Philadelphia PA: American Society for Testing and Materials.

Athanasopoulos, G. A. (2008).” Laboratory testing of municipal solid waste”. GEOTECHNICAL SPECIAL PUBLICATION, 209, 196.

Attal, A., Akunna, J., Camacho, P., Salmon, P., and Paris, I. (1992). "Anaerobic degradation of municipal wastes in landfill." Water Science & Technology, 25(7), 243-253.

Bareither, C. A., Breitmeyer, R. J., Erses, A. S., Benson, C. H., Edil, T. B., & Barlaz, M. A. (2008). “Relative contributions of moisture and biological activity on compression of municipal solid waste in bioreactor landfills”. In Proceedings of the Global Waste Symposium 2008.

Barlaz, M. A. (2006). "Forest products decomposition in municipal solid waste landfills." Waste Manage. 26(4), 321-333.

Barlaz, M. A., Bareither, C. A., Hossain, A., Saquing, J., Mezzari, I., Benson, C. H., & Yazdani, R. (2010). “Performance of North American bioreactor landfills. II: Chemical and biological characteristics”. Journal of Environmental Engineering, 136(8), 839-853.

Barlaz, M. A., Ham, R. K., & Schaefer, D. M. (1989). “Mass-balance analysis of anaerobically decomposed refuse”. Journal of Environmental Engineering, 115(6), 1088-1102.

Barlaz, M. A., Ham, R. K., & Schaefer, D. M. (1992). “Microbial, chemical and methane production characteristics of anaerobically decomposed refuse with and without leachate recycling”. Waste Management & Research, 10(3), 257-267.

Barlaz, M. A., Ham, R. K., Schaefer, D. M., & Isaacson, R. (1990). “Methane production from municipal refuse: a review of enhancement techniques and microbial dynamics”. Critical Reviews in Environmental Science and Technology, 19 (6), 557-584.

Barlaz, M. A., Milke, M. W., & Ham, R. K. (1987). “Gas production parameters in sanitary landfill simulators”. Waste Management & Research, 5(1), 27-39.

Barlaz, M., Eleazer, W., Odle Iii, W., Qian, X., and Wang, Y. (1997). "Biodegradative analysis of municipal solid waste in laboratory-scale landfills" Environmental Protection Agency, Cincinnati, OH.

Beaven, R. P. (1996). “Determination of the hydrogeological and geotechnical properties of refuse in relation to sustainable landfilling”. In Proceedings of the 19th international Madison waste conference, University of Wisconsin-Madison (pp. 435-454). University of Wisconsin-Madison.

Beaven, R. P. (2000). “The hydrogeological and geotechnical properties of household waste in relation to sustainable landfilling” (Doctoral dissertation, University of Southampton, U.K.).

Beaven, R. P., Powrie, W., & Zardava, K. (2011). “Hydraulic properties of MSW”. Geotechnical characterization, Field Measurements and Laboratory Testing of Municipal Solid Waste. Virginia, US, American Society of Civil Engineers, 1-43.

Beaven, R.P. and Powrie, W. (1995). “Hydrological and geotechnical properties of refuse using a large scale compression cell.” Proceedings of Fifth International landfill Symposium, Sardinia, Cagliari, volume II: 745-760.

Beaven, R.P. and Powrie, W. (1995). “Hydrological and geotechnical properties of refuse using a large scale compression cell.” Proceedings of Fifth International landfill Symposium, Sardinia, Cagliari, volume II: 745-760.

Beker, I. D (1987). Control of acid degradation. In: Christensen, T.H., Cossu, R. and Stegmann, R. (eds). Proceeding Sardinia 87’, First International landfill Symposium. Volume 2. Cagliari, Italy: CISA. Pp. 1-22. Cited in Komilis et ah, (1999a)

Benson, C. H., Thorstad, P. A., Jo, H. Y., & Rock, S. A. (2007). “Hydraulic performance of geosynthetic clay liners in a landfill final cover”. Journal of Geotechnical and Geoenvironmental Engineering, 133(7), 814-827.

Bilgili, M. S., Demir, A., & Özkaya, B. (2007). “Influence of leachate recirculation on aerobic and anaerobic decomposition of solid wastes”. Journal of hazardous materials, 143(1), 177-183.

Bingemer, H. G., & Crutzen, P. J. (1987). “The production of methane from solid wastes” Journal of Geophysical Research: Atmospheres (1984–2012), 92(D2), 2181-2187.

Bleiker, D. E., Farquhar, G., & McBean, E. (1995). “Landfill settlement and the impact on site capacity and refuse hydraulic conductivity”. Waste management & research, 13(6), 533-554.

Bleiker, E.D., McBean, E., and Farquhar, G. (1993). “Refuse Sampling and Permeability Testing At The Brock Westand Keele Valley Landfills” Proceedings, Sixteenth International Madison Waste Conference: municipal & industrial waste, September 22-23, 1993. 548-567.

Bogner, J. E. (1990). "Controlled Study of Landfill Biodegradation Rates Using Modified Bmp Assays." Waste Management Research, 8(1), 329-352.

Bookter, T. J., and Ham, R. K. (1982). "Stabilization of solid waste in landfills." Journal of the Environmental Engineering Division, 108(6), 1089-1100.

Brandl, H. (1994). “Vertical barriers for municipal and hazardous waste containment”. In Proceedings Symposium on Developments in geotechnical Engineering, Bangkok (301-334).

Brenda, B., Calvert, P. P., Pettigrew, C. A., and Barlaz, M. A. (1998). "Testing anaerobic biodegradability of polymers in a laboratory-scale simulated landfill." Environ. sci.Technol.32 (6), 821-827.

Brown, K. A., Smith, A., Burnley, S. J., Campbell, D. J. V., King, K., and Milton, M. J. T. (1999). "Methane Emissions from UK Landfills." Rep. No. AEAT-5217, AEA Technology Environment, Oxfordshire, UK.

Buivid, M. G., Wise, D. L., Blanchet, M. J., Remedios, E. C., Jenkins, B. M., Boyd, W. F., & Pacey, J. G. (1981). “Fuel gas enhancement by controlled landfilling of municipal solid waste”. Resources and Conservation, 6(1), 3-20.

Chan, G. Y. S., Chu, L. M., & Wong, M. H. (2002). “Effects of leachate recirculation on biogas production from landfill co-disposal of municipal solid waste, sewage sludge and marine sediment”. Environmental Pollution, 118(3), 393-399.

Chanton, J. P., Powelson, D.K., Green R. B., (2009). "Methane Oxidation in Landfill Cover Soils: Is a 10% Default Value Resonable?" Journal of Environmental Quality, 38 (2), 654-663.

Chen, J., Wang, H., and Zhang, N. (2009). "Modified landfill gas generation model of first order kinetics and two stage reaction." Environmental Science and Engineering (China), 3(3), 313-319.

Chen, T.H. and Chynoweth, D.P. (1995). “Hydraulic conductivity of compacted municipal solid waste.” Bioresource Technology 51(2-3): 205-212.

Chen, Y. M., Zhan, T. L., Wei, H. Y., & Ke, H. (2009). “Aging and compressibility of municipal solid wastes”. Waste management, 29(1), 86-95.

Chiampo, F., Conti, R., and Cometto, D. (1996). "Morphological characterisation of MSW landfills." Resour. Conserv. Recycling, 17(1), 37-45.

Chiemchaisri, C., Chiemchaisri, W., Kumar, S., & Hettiaratchi, J. P. A. (2007). “Solid waste characteristics and their relationship to gas production in tropical landfill.” Environmental monitoring and assessment, 135(1-3), 41-48.

Cho, J. K., Park, S. C., and Chang, H. N. (1995). "Biochemical methane potential and solid state anaerobic digestion of Korean food wastes." Bioresour. Technol. 52(3), 245-253.

Chugh, S., Chynoweth, D., Clarke, W., Pullammanappallil, P. & Rudolph V. (1998). “Effect of recirculated leachate volume on MSW degradation. Waste Management” Research, Vol. 16, 564-573.

Christensen, A. H., & Quail, P. H. (1996). “Ubiquitin promoter-based vectors for high-level expression of selectable and/or screenable marker genes in monocotyledonous plants”. Transgenic research, 5(3), 213-218.

Christensen, T. H., and Kjeldsen, P. (1989). "Basic Biochemical Processes in Landfills." Sanitary Landfilling: Process, Technology, and Environmental Impact, 29-49.

Chynoweth, D., Turick, C., Owens, J., Jerger, D., and Peck, M. (1993). "Biochemical methane potential of biomass and waste feedstocks." Biomass Bioenergy, 5(1), 95-111.

Cooper, C. D., Reinhart, D., Rash, F., Seligman, D., and Keely, D. (1992). "Landfill gas emissions." Florida Center for Solid and Hazardous Wastes Management, Report, 92-92.

Cruz, F. B. D., and Barlaz, M. A. (2010). "Estimation of waste component-specific landfill decay rates using laboratory-scale decomposition data." Environ. Sci. Technol., 44(12), 4722- 4728.

Dean, A. M., and Voss, D. (1999). Design and analysis of experiments. Springer Verlag,

Dixon, N., & Jones, D. R. V. (2005).”Engineering properties of municipal solid waste”. Geotextiles and Geomembranes, 23(3), 205-233.

Durmusoglu, E., Sanchez, M. Itza and Corapcioglu, Y. M. (2006). “Permeability and compression characteristics of Municipal solid waste” Environ Geol. 50, 773-789

Eggleston, H. S., Buendia, L., Miwa, L., and Ngara, T. (2006). "Intergovernmental Panel of Climate Change (IPCC) Guidelines for National Greenhouse Inventories." Tanabe, IGES, Japan.

Eleazer, W. E., Odle III, W. S., Wang, Y. S., and Barlaz, M. A. (1997). "Biodegradability of municipal solid waste components in laboratory-scale landfills." Environ. Sci. Technol., 31(3), 911-917.

El-Fadel, M. (1999). “Simulating temperature variations in landfills”. Journal of Solid Waste Technology and Management, 26(2), 78-86.

El-Fadel, M., and Al-Rashed, H. (1998). ‘‘Settlement in municipal solid waste landfills I. Field scale experiments.’’ J. Solid Waste Technol. Manage, 25(2), 89–98.

El-Fadel, M., Findikakis, A. N., & Leckie, J. O. (1996). “Numerical modelling of generation and transport of gas and heat in landfills I. Model formulation”. Waste management & research, 14(5), 483-504.

El-Fadel, M., Findikakis, A. N., & Leckie, J. O. (1997). “Modeling leachate generation and transport in solid waste landfills”. Environmental technology, 18(7), 669-686. Eng., 1104, 780–796.

Erses, A. S., Onay, T. T., & Yenigun, O. (2008). “Comparison of aerobic and anaerobic degradation of municipal solid waste in bioreactor landfills”. Bioresource Technology, 99(13), 5418-5426.

Erses, S., Onay, T.T.,(2003).” In situ heavy metal attenuation in landfills under methanogenic conditions”. Journal of Hazardous Materials 99, 159–175.

Ettala, M. (1987). “Infiltration and hydraulic conductivity at a sanitary landfill.” Aqua Fenn., 17, 231–237.

Faour, A. A., Reinhart, D. R., and You, H. (2007). "First-order kinetic gas generation model parameters for wet landfills." Waste Manage, 27(7), 946-953.

Farquhar, G., and Rovers, F. (1973). "Gas production during refuse decomposition." Water, Air, & Soil Pollution, 2(4), 483-495.

Fassett, J. B., Leonards, G. A., & Repetto, P. C. (1994). “Geotechnical properties of municipal solid wastes and their use in landfill design”. Waste Tech, 94, 13-14.

Filipkowska, U., and Agopsowicz, M. (2004). "Solids waste gas recovery under different water conditions." Polish Journal of Environmental Studies, 13(6), 663-669.

Francois, V., Feuillade, G., Skhiri, N., Lagier, T., and Matejka, G. (2006). "Indicating the parameters of the state of degradation of municipal solid waste." J. Hazard. Mater., 137(2), 1008-1015.

Fredlund, D. G., & Rahardjo, H. (1993). “Soil mechanics for unsaturated soils”. John Wiley & Sons.

Fungaroli, A. A., & Steiner, R. L. (1979). Investigation of sanitary landfill behavior; final report. In Investigation of sanitary landfill behavior; final report. EPA.

Gabr, M. A., & Valero, S. N. (1995). “Geotechnical properties of municipal solid waste”. ASTM geotechnical testing journal, 18(2), 241-251.

Godley, A., Lewin, K., Graham, A., and Smith, R. (2003). "Environment agency review of methods for determining organic waste biodegradability for landfill and municipal waste diversion." Proceedings 8th European Biosolids and Organic Residuals Conference, Wakefield, UK, 14.

Gourc, J.P., Olivier, F., Thomas, S., Chatelet, L., Denecheau, P., Munoz, M.L.,(2001). “Monitoring of waste settlements on five landfills: comparison of the efficiency of different devices”. Proceedings Sardinia ’01, Eighth International Waste Management and Landfill Symposium, Cagliari, Italy, pp. 515–524.

Grellier, S., Reddy, K. R., Gangathulasi, J., Adib, R., and Peters, C. (2007). “U.S. MSW and its biodegradation in a bioreactor landfill.” Proc., Sardinia 2007, 11th Int. Landfill Symp., S. Margherita di Pula, CISA, Environmental Sanitary Engineering Center, Cagliari, Italy.

Guermoud, N., Ouadjnia, F., Abdelmalek, F., & Taleb, F. (2009). “Municipal solid waste in Mostaganem city” (Western Algeria). Waste Management, 29(2), 896-902.

Gunaseelan, V. N. (2004). "Biochemical methane potential of fruits and vegetable solid waste feedstocks." Biomass Bioenergy, 26(4), 389-399.

Gurijala, K. R., and Suflita, J. M. (1993). "Environmental factors influencing methanogenesis from refuse in landfill samples." Environ. Sci. Technol., 27(6), 1176-1181.

Gurijala, K., Sa, P., and Robinson, J. (1997). "Statistical Modeling of Methane Production from Landfill Samples." Appl. Environ. Microbial., 63(10), 3797-3803.

Ham, R. K. (1979). “Recovery, processing, and utilization of gas from sanitary landfills”. Environmental Protection Agency, Office of Research and Development, Municipal Environmental Research Laboratory.

Ham, R. K., & Bookter, T. J. (1982). “Decomposition of solid waste in test lysimeters”. Journal of the Environmental Engineering Division, 108(EE6), 1147-70.

Ham, R. K., Reinhardt, J. J., and Sevick, G. W. (1978). “Density of milled and unprocessed refuse.” J. Envir. Eng. Div., 104EE1, 109–125.

Hansen, T. L., Schmidt, J. E., Angelidaki, I., Marca, E., Jansen, J. C., Mosbæk, H., and Christensen, T. H. (2004). "Method for determination of methane potentials of solid organic waste." Waste Manage. 24(4), 393-400.

Hanson, J. L., Yeşiller, N., and Oettle, N. K. (2010). "Spatial and Temporal Temperature Distributions in Municipal Solid Waste Landfills." J. Environ. Eng., 136 804.

Hanson, J. L., Yesiller, N., Von Stockhausen, S. A., & Wong, W. W. (2010). Compaction characteristics of municipal solid waste. Journal of geotechnical and geoenvironmental engineering, 136(8), 1095-1102.

Hao, Y. J., Wu, W. X., Wu, S. W., Sun, H., & Chen, Y. X. (2008). Municipal solid waste decomposition under oversaturated condition in comparison with leachate recirculation. Process Biochemistry, 43(1), 108-112.

Haque, M. A. (2007). “Dynamic characteristics and stability analysis of municipal solid waste in bioreactor landfills”. PhD dissertation, The University of Texas at Arlington.

Harris, M. R. R. (1979). “Geotechnical characteristics of landfilled domestic refuse.” The Engineering behavior of Industrial and Urban Fill: Proc. of the Symposium held at the Univ. of Birmingham, Midland

Hartz, K. E., Klenk, R. E., & Ham, R. K. (1982). “Temperature effects: methane generation from landfill samples”. Journal of the Environmental Engineering Division, 108(EE4), 629-38.

Haydar, M. M., & Khire, M. V. (2005). “Leachate recirculation using horizontal trenches in bioreactor landfills”. Journal of geotechnical and geoenvironmental engineering, 131(7), 837-847.

Hernández-Berriel, M. C., Mañón-Salas, C., Sánchez-Yáñez, J., Lugo-de la Fuente, J., and Márquez-Benavides, L. (2010). "Influence of Recycling Different Leachate Volumes on Refuse Anaerobic Degradation." The Open Waste Management Journal, 155-166

Hettiarachchi, C. H. (2005). “Mechanics of biocell landfill settlements”. Ph.D. dissertation, New Jersey Institute of Technology, Newark, N.J.

Hillel, D. (1971). Soil and water: physical principles and processes.

Hossain, M. S., and Gabr, M. A., (2009). “The effect of shredding and test apparatus size on compressibility and strength parameters of degraded municipal solid waste”. Waste management, 29(9), 2417-2424.

Hossain, M. S., Penmethsa, K. K., & Hoyos, L. (2009). “Permeability of municipal solid waste in bioreactor landfill with degradation”. Geotechnical and Geological Engineering, 27(1), 43-51.

Hudson A.P., Beaven R.P. and Powrie W. (2001). “Interaction of water and gas in saturated household waste in a large scale compression cell.” Proceedings of the 8th International Sardinia Landfill Conference, S. Margherita Dipula, Cagliari, Italy. Vol. III: 585-594.

Hudson, A. P., White, J. K., Beaven, R. P., & Powrie, W. (2004). “Modelling the compression behavior of landfilled domestic waste”. Waste Management, 24(3), 259-269.

Hyder, M.M., Khire, M.V. (2004) “Evaluation of heterogeneity and anisotropy of waste properties on leachate recirculation in bioreactor landfills”. J Solid Waste Technol Manage, 30(4): 233–242

Intergovernmental Panel on Climate Change (IPCC), 2006, “IPCC Guidelines for National Greenhouse Gas Inventories”, Eggleston H.S., Buendia L., Miwa K., Ngara T. and Tanabe K. (eds), Institute for Global Environmental Strategies, Japan.

Intergovernmental Panel on Climate Change (IPCC), 2007, “Changes in Atmospheric Constituents and in Radiative Forcing. The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report” Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Isci, A., & Demirer, G. N. (2007). Biogas production potential from cotton wastes. Renewable energy, 32(5), 750-757.

Ivanova, L. K., Richards, D. J., and Smallman, D. J. (2008). "Assessment of the anaerobic biodegradation potential of MSW." Proceedings of the Institution of Civil Engineers-Waste and Resource Management, 167-180.

Jain, P., Powell, J., Townsend, T. G., and Reinhart, D. R. (2006). “Estimating the hydraulic conductivity of landfilled municipal solid waste using borehole permeameter test.” J. Environ. Eng., 132(6), 645–653.

Jang, Y. S., Kim, Y. W., and Lee, S. I. (2002). “Hydraulic properties and leachate level analysis of Kimpo metropolitan landfill, Korea.” Waste Manage. 22, 261–267.

Jeon, E., Bae, S., Lee, D., Seo, D., Chun, S., Lee, N., and Kim, J. (2007). "Methane generation potential and biodegradability of MSW components." Sardinia 2007 Eleventh International Waste Management and Landfill Symposium.

Jones, H. A., and Nedwell, D. B. (1993). "Methane emission and methane oxidation in landfill cover soil." FEMS Microbiol.Lett., 102(3-4), 185-195.

Jones, K., Rees, J., and Grainger, J. (1983). "Methane generation and microbial activity in a domestic refuse landfill site." Appl. Microbiol. Biotechnol., 18(4), 242-245.

Kamalan, H., Sabour, M., and Shariatmadari, N. (2011). "A review on available landfill gas models." Journal of Environmental Science and Technology, 4(2), 79-92.

Karanjekar, R. V. (2013). “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.

Kasali, G. B., & Senior, E. (1989). “Effects of temperature and moisture on the anaerobic digestion of refuse”. Journal of Chemical Technology and Biotechnology, 44(1), 31-41.

Kelly, R. J., Shearer, B. D., Kim, J., Goldsmith, C. D., Hater, G. R., and Novak, J. T. (2006). "Relationships between analytical methods utilized as tools in the evaluation of landfill waste stability." Waste Manage., 26(12), 1349-1356.

Khire, M. V., & Mukherjee, M. (2007). “Leachate injection using vertical wells in bioreactor landfills.” Waste Management, 27(9), 1233-1247.

Kim, J., & Pohland, F. (2003). “Process enhancement in anaerobic bioreactor landfills”. Water Science & Technology, 48(4), 29-36.

Kjeldsen, P., Barlaz, M. A., Rooker, A. P., Baun, A., Ledin, A., & Christensen, T. H. (2002). “Present and long-term composition of MSW landfill leachate: a review.” Critical reviews in environmental science and technology, 32(4), 297-336.

Klink, R. E., & Ham, R. K. (1982). “Effects of moisture movement on methane production in solid waste landfill samples”. Resources and Conservation, 8(1), 29-41.

Koerner, R.M., and Soong, T-Y., 2000. “Leachate in landfills: the stability issues”. Geotextiles and Geomembranes 18(5), 293–309.

Komilis, D. P., and Ham, R. K. (2003). "The effect of lignin and sugars to the aerobic decomposition of solid wastes." Waste Manage. 23(5), 419-423.

Korfiatis, G. P., Demetracopoulos, A. C., Bourodimos, E. L., & Nawy, E. G. (1984). “Moisture transport in a solid waste column”. Journal of Environmental Engineering, 110(4), 780-796.

Landva, A., & Knowles, G. D. (Eds.). (1990). Geotechnics of waste fills: theory and practice (No. 1070). ASTM International.

Landva, A. O., and Clark, J. I. (1986). “Geotechnical testing of waste fill,” Proc., 39th Canadian Geotechnical Conf., Canadian Geotechnical Society, Ottawa, Ont., Canada, 371–385.

Landva, A. O., Clark, J. I., Weisner, W. R., & Burwash, W. J. (1984). Geotechnical engineering and refuse landfills. In Proceedings of the 6th National Conference on Waste Management in Canada, Vancouver, BC (pp. 1-37).

Landva, A. O., Pelkey, S. G., & Valsangkar, A. J. (1998). “Coefficient of permeability of municipal refuse”. In Proceedings of the 3rd International Congress on Environmental Geotechnics, Lisbon (Vol. 1, pp. 63-68).

Landva, A. O., Valsangkar, A. J., & Pelkey, S. G. (2000). “Lateral earth pressure at rest and compressibility of municipal solid waste”. Canadian Geotechnical Journal, 37(6), 1157-1165.

Lay, J. J., Li, Y. Y., and Noike, T. (1998). "Developments of bacterial population and methanogenic activity in a laboratory-scale landfill bioreactor." Water Res., 32(12), 3673- 3679.

Leckie, J. O., Halvadakis, C., & Pacey, J. G. (1979). “Landfill management with moisture control”. Journal of the Environmental Engineering Division, 105(2), 337-355.

Leuschner, A. (1982). “Enhancement of degradation: Laboratory- scale experiments”. In sanitary landfilling process, Technology and Environmental Impact, Christensen,T.H., Cossu R. and Stegmann R. eds, Academic Press, London, UK, 83- 102.

Liu, J. L., Ke, H., Zhan, T. L., & Chen, Y. M. (2010). “Simulation tests of biodegradation and compression of municipal solid waste” In Advances in Environmental Geotechnics (pp. 521-524). Springer Berlin Heidelberg.

Machado, S. L., Carvalho, M. F., Gourc, J. P., Vilar, O. M., and Do Nascimento, J. C. F. (2009). "Methane generation in tropical landfills: Simplified methods and field results." Waste Manage., 29(1), 153-161.

Manassero, M., Van Impe, W. F., & Bouazza, A. (1996). “Waste disposal and containment”-State of the art lecture, 193-242.

Mason, R. L., Gunst, R. F., and Hess, J. L. (1989). "Statistical design and analysis of experiments." New York, Wiley.

Mata-Alvarez, J., & Martinez-Viturtia, A. (1986). “Laboratory simulation of municipal solid waste fermentation with leachate recycle”. Journal of Chemical Technology and Biotechnology, 36(12), 547-556.

Maurice, C., and Lagerkvist, A. (1997). "Seasonal variation of landfill gas emissions." Proceedings of Sixth International Landfill Symposium in Sardinia, Caliagri,Italy, 87-93.

McCreanor, P. T., & Reinhart, D. R. (1999). “Hydrodynamic modeling of leachate recirculating landfills”. Waste management and research, 17(6), 465-469.

McDougall, J. R., Pyrah, I. C., & Yuen, S. T. S. (2004). “Extended phase relations and load effects in MSW”. Waste Management, 24(3), 251-257.

Mehta, R., Barlaz, M., Yazdani, R., Augenstein, D., Bryars, M., & Sinderson, L. (2002). "Refuse Decomposition in the Presence and Absence of Leachate Recirculation." Journal of Environmental Engineering, 228-236.

Meima, J., Naranjo, N. M., and Haarstrick, A. (2008). "Sensitivity analysis and literature review of parameters controlling local biodegradation processes in municipal solid waste landfills." Waste Manage, 28(5), 904-918.

Murphy, R. J., Jones, D. E., & Stessel, R. I. (1995). “Relationship of microbial mass and activity in biodegradation of solid waste”. Waste management & research, 13(5), 485-497.

Olivier, F., & Gourc, J. P. (2007). “Hydro-mechanical behavior of Municipal Solid Waste subject to leachate recirculation in a large-scale compression reactor cell”. Waste Management, 27(1), 44-58.

Oonk, H. (2010). “Literature review: methane from landfills, methods to quantify generation, oxidation and emission”. Innovations in Environmental Technology.

Oonk, H., and Boom, T. (1995a). "Landfill Gas Formation, Recovery and Emissions." Rep. No. TNO-report R 95-203, Appledorm, Netherlands.

Oonk, H., and Boom, T. (1995b). "Validation of landfill gas formation models." Studies in Environmental Science, 65 597-602.

Otieno, F. O., 1989. Leachate recirculation in landfills as a management technique. In Proceedings of Sardinia'89, Second Int. Landfill Symposium, Calgari, Italy.

Oweis, I. S., & Khera, R. P. (1990). "Geotechnology of waste management".

Oweis, I. S., Smith, D. A., Brian Ellwood, R., & Greene, D. S. (1990). "Hydraulic characteristics of municipal refuse". Journal of Geotechnical Engineering, 116(4), 539-553.

Owens, J., and Chynoweth, D. (1993). "Biochemical methane potential of municipal solid waste (MSW) components." Water Science & Technology, 27(2), 1-14.

Pacey, J. (1989). "Enhancement of degradation: Large-scale experiments. IN: Sanitary Landfilling: Process, Technology, and Environmental Impact". Academic Press, New York.

Pacey, J., Augenstein, D., Morck, R., Reinhart, D., & Yazdani, R. (1999). "The bioreactor landfill-an innovation in solid waste management". MSW management, 53-60.

Penmethsa, K. K. (2007). "Permeability of municipal solid waste in bioreactor landfill with degradation." MS thesis, Univ. of Texas at Arlington, Arlington, Tex.

Pohland, F. G. (1975). ''Sanitary landfill stabilization with leachate recycle and residential treatment.'' EPA Grant No. R-801397, U.S.E.P.A. National Environmental Research Center, Cincinnati.

Pohland, F. G. (1995). "Landfill bioreactors: historical perspective, fundamental principles and new horizons in design and operations". Landfill Bioreactor Design and Operation Sem. Proc., EPA/600/R-95/146, 9-24.

Pohland, F. G. (1996). "Landfill bioreactors: fundamentals and practice". Water quality international, 1996(9-10), 18-22.

Pohland, F. G., & Harper, S. R. (1986). "Critical Review and Summary of Leachate and Gas Production from Landfills." EPA/600/2-86/073, OH, USA.

Pohland, F., & Al-Yousfi, B. (1994). “Design and operation of landfills for optimum stabilization and biogas production”. Water Science and Technology, 30(12), 117-124.

Pohland, F., & Kim, J. (2000). “Microbially mediated attenuation potential of landfill bioreactor systems”. Water Science and Technology, 41(3), 247-254.

Powrie, W., & Beaven, R. P. (1999). “Hydraulic properties of household waste and implications for landfills”. Proceedings of the ICE-Geotechnical Engineering, 137(4), 235-247.

Powrie, W., Beaven R.P. and Hudson A.P. (2005) “Factors affecting the hydraulic conductivity of waste” In: International Workshop "Hydro-physico-mechanics of landfills." Ed. Gourc J.P., Grenoble (2005).

Powrie, W., Beaven, R. P., & Hudson, A. (2008). “The influence of landfill gas on the hydraulic conductivity of waste”. Geotechnical Special Publication, (177), 264-271.

Powrie, W., Richards, D. J., & Beaven, R. P. (1998). “Compression of waste and implications for practice”. Geotechnical Engineering of Landfills, Thomas Telford, 3-18.

Qudais M., Qdais H.A. (2000). "Energy Content of Municipal Solid Waste in Jordan and its Potential Utilization". Energy Conversion and Management, 41, 983-991

Radcliffe, D. E., & Šimůnek, J. (2010). “Soil physics with HYDRUS: modeling and applications”. Boca Raton, FL: CRC press.

Rao, M., Singh, S., Singh, A., and Sodha, M. (2000). "Bioenergy conversion studies of the organic fraction of MSW: assessment of ultimate bioenergy production potential of municipal garbage." Appl. Energy, 66(1), 75-87.

Reddy, K. R., Hettiarachchi, H., Parakalla, N. S., Gangathulasi, J., & Bogner, J. E. (2009, b). “Geotechnical properties of fresh municipal solid waste at Orchard Hills Landfill, USA”. Waste Management, 29(2), 952-959.

Reddy, K. R., Hettiarachchi, H., Parakalla, N., Gangathulasi, J., Bogner, J., & Lagier, T. (2009, a). “Hydraulic conductivity of MSW in landfills”. Journal of Environmental Engineering, 135(8), 677-683.

Rees, J. F. (1980). "Optimisation of methane production and refuse decomposition in landfills by temperature control." Journal of Chemical Technology and Biotechnology, 30(1), 458-465.

Reinhart, D. R. (1996). “Full-scale experiences with leachate recirculating landfills: case studies”. Waste Management & Research, 14(4), 347-365.

Reinhart, D. R., & Al-Yousfi, A. B. (1996). “The impact of leachate recirculation on municipal solid waste landfill operating characteristics”. Waste Management & Research, 14(4), 337-346.

Reinhart, D. R., & Townsend, T. G. (1997). “Landfill bioreactor design & operation”. CRC press.

Reinhart, D. R., Faour, A. A., You, H., National Risk Management Research Laboratory (US). Air Pollution Prevention and Control Division, and University of Central Florida. (2005). “First- Order Kinetic Gas Generation Model Parameters for Wet Landfills”. US Environmental Protection Agency, Office of Research and Development, Washington D.C.

Reinhart, D. R., McCreanor, P. T., & Townsend, T. (2002). “The bioreactor landfill: its status and future”. Waste Management & Research, 20 (2), 172-186.

Rendra S., Fernandes L. & Warith M. (2003). “Degradation of municipal solid waste in simulated aerobic and anaerobic bioreactor landfills”. Journal of Solid Waste Technology and Management, Vol. 29, n3, 145- 153.

Ress, B. B., Calvert, P. P., Pettigrew, C. A., and Barlaz, M. A. (1998). "Testing Anaerobic Biodegradability of Polymers in a Laboratory-Scale Simulated Landfill." Environmental Science & Technology, 32(6), 821-827.

Rhew, R. D., & Barlaz, M. A. (1995).”Effect of lime-stabilized sludge as landfill cover on refuse decomposition”. Journal of environmental engineering, 121(7), 499-506.

Rowe, R. K., & Nadarajah, P. (1996). “Estimating leachate drawdown due to pumping wells in landfills”. Canadian geotechnical journal, 33(1), 1-10.

Šan, I., & Onay, T. T. (2001). “Impact of various leachate recirculation regimes on municipal solid waste degradation”. Journal of Hazardous Materials, 87(1), 259-271.

Sanphoti, N., Towprayoon, S., Chaiprasert, P., and Nopharatana, A. (2006). "The effects of leachate recirculation with supplemental water addition on methane production and waste decomposition in a simulated tropical landfill." J. Environ. Manage., 81(1), 27-35.

Scharff, H., and Jacobs, J. (2006). "Applying guidance for methane emission estimation for landfills." Waste Manage., 26(4), 417-429.

Senior, E., & Kasali, G. B. (1990). “Landfill gas. Microbiology of landfill sites”. CRC Press, Boca Raton, Fla, 113-158.

Shank, K. L. (1993). “Determination of the hydraulic conductivity of the Alachua County southwest landfill”. NAVAL POSTGRADUATE SCHOOL MONTEREY CA.

Shanmugam, P., and Horan, N. (2009). "Simple and rapid methods to evaluate methane potential and biomass yield for a range of mixed solid wastes." Bioresour. Technol., 100(1), 471-474.

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,17 (3), 371-374

Sharma, H. D., and Reddy, K. R. (2004). “Geoenvironmental engineering: Site remediation, waste containment, and emerging waste management technologies”, Wiley, Hoboken, N.J.

Shelton, D. R., and Tiedje, J. M. (1984). "General method for determining anaerobic biodegradation potential." Appl. Environ. Microbial., 47(4), 850.

Sivanesan, Y. (2013). “Effect of saline water on landfill gas generation of municipal solid waste”. Master Thesis, University of Texas at Arlington

Solid Waste Association of North America. (1997). "Comparison of Models for Predicting Landfill Methane Recovery." Rep. No. GR-LG0075, SWANA, Maryland.

Solomon, S. (Ed.). (2007). “Climate change 2007-the physical science basis”: Working group I contribution to the fourth assessment report of the IPCC (Vol. 4). Cambridge University Press.

Spokas, K., Bogner, J., Chanton, J., & Franco, G. (2009, October). “Developing a new field-validated methodology for landfill methane emissions in California”. In Proc. Sardinia (Vol. 9).

Sponza, D. T., & Ağdağ, O. N. (2005). “Effects of shredding of wastes on the treatment of municipal solid wastes (MSWs) in simulated anaerobic recycled reactors”. Enzyme and Microbial Technology, 36(1), 25-33

Staley, B. F., and Barlaz, M. A. (2009). "Composition of municipal solid waste in the united states and implications for carbon sequestration and methane yield." J. Environ. Eng., 135 901.

Staub, M., Galietti, B., Oxarango, L., Khire, M. V., & Gourc, J. P. (2009, March). Porosity and hydraulic conductivity of MSW using laboratory-scale tests. In Third International Workshop" Hydro-Physico-Mechanics of Landfills", Braunschweig, Germany (pp. 10-13).

Stege, G.A. (2010). "User's Model Columbia's Gas Model- Version 1.0.", US EPA's Landfill Methane Outreach Program, U.S. EPA, Washington D.C.

Stegman R. (1983) “New aspects on enhancing biological processes in sanitary landfill”. Waste Management and Research, 1, 201- 211.

Stoltz, G., Gourc, J. P., & Oxarango, L. (2010a). “Characterisation of the physico-mechanical parameters of MSW”. Waste Management, 30(8), 1439-1449.

Stoltz, G., Gourc, J. P., & Oxarango, L. (2010b). “Liquid and gas permeabilities of unsaturated municipal solid waste under compression”. Journal of contaminant hydrology, 118(1), 27-42.

Stone, R., and Kahle, R. L. (1972). "Water and Sewage Sludge Absorption by Solid Waste." Journal of the Sanitary Engineering Division, 98(5), 731-743.

Suler, D. J., & Finstein, M. S. (1977). “Effect of temperature, aeration, and moisture on CO_2 formation in bench-scale, continuously thermophilic composting of solid waste”. Applied and Environmental Microbiology, 33(2), 345-350.

Tchbanoglaus, G., Theisen, H., & Vigil, S. (1993). “Integrated Solid Waste Management”. New York, NY, McGraw-Hill.

Themelis, N. J., & Ulloa, P. A. (2007). “Methane generation in landfills”. Renewable Energy, 32(7), 1243-1257.

Thompson, S., Sawyer, J., Bonam, R., and Valdivia, J. (2009). "Building a better methane generation model: Validating models with methane recovery rates from 35 Canadian landfills." Waste Manage. 29(7), 2085-2091.

Tolaymat, T. M., Green, R. B., Hater, G. R., Bariaz, M. A., Black, P., Branson, D., and Powell, J. (2010). "Evaluation of landfill gas decay constant for municipal solid waste landfills operated as bioreactors." J. Air Waste Manage. Assoc., 60(1), 2187.

Townsend, T. G., Miller, W. L., Lee, H. J., & Earle, J. F. K. (1996). “Acceleration of landfill stabilization using leachate recycle”. Journal of Environmental Engineering, 122(4), 263-268.

United States Environmental Protection Agency (USEPA) (February 2014). "Municipal Solid Waste Generation, Recycling, and Disposal in the United States: Fact and Figures for 2012." http://www.epa.gov/solidwaste/nonhaz/municipal/msw99.htm

United States Environmental Protection Agency (USEPA), (1998). "Fate and Transport and Transformation Test Guidelines OPPTS 835.3400 Anaerobic Biodegradability of Organic Chemicals." Rep. No. EPA 712-C-98-090.

United States Environmental Protection Agency (USEPA), 2005, "Landfill Gas Emissions Model (LandGEM) Version 3.02 User's Guide", Report no. EPA-600/R-05/047, U.S. Environmental Protection Agency, Office of Research and Development, Washington, DC

United States Environmental Protection Agency (USEPA), 2011, “U.S. Greenhouse Gas Inventory Report: Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2009”, Report No. EPA 430-R-11-005, USEPA, Washington, DC

United States Environmental Protection Agency. Office of Research and Development. (2005). “Landfill gas emissions model” (LandGEM) version 3.02 user's guide. Washington D.C.

Valencia, R., Van der Zon, W., Woelders, V. D. H., Lubberding, H. J., & Gijzen, H. J. (2009). “The effect of hydraulic conditions on waste stabilization in bioreactor landfill simulators”. Bioresource technology, 100(5), 1754-1761.

Vavilin, V., Lokshina, L. Y., Jokela, J., and Rintala, J. (2004). "Modeling solid waste decomposition." Bioresour. Technol. 94(1), 69-81.

Wang, Y. S., Byrd, C. S., and Barlaz, M. A. (1994). "Anaerobic biodegradability of cellulose and hemicellulose in excavated refuse samples using a biochemical methane potential assay." J. Ind. Microbiol. Biotechnol., 13(3), 147-153.

Wang, Y. S., Odle, W. S., Eleazer, W. E., and Bariaz, M. A. (1997). "Methane potential of food waste and anaerobic toxicity of leachate produced during food waste decomposition." Waste Manage. Res., 15(2), 149-167.

Wang-Yao, K., Towprayoon, S., Chiemchaisri, C., Gheewala, S. H., and Nopharatana, A. (2006). "Seasonal variation of landfill methane emissions from seven solid waste disposal sites in central Thailand." Proc. of the 2nd Joint International Conference on ‘Sustainable Energy and Environment’ (SEE 2006), 21-23.

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

Warith, M. A. (2003). Solid Waste Management: New trends in landfill design. Emirates Journal of Engineering Research, Vol. 8, No. 1: 61-70.

Warith, M. A., & Sharma, R. (1998). “Technical review of methods to enhance biological degradation in sanitary landfills”. Water Quality Research Journal of Canada, 33(3), 417-437.

Warith, M. A., Zekry, W., & Gawri, N. (1999). “Effect of leachate recirculation on municipal solid waste biodegradation”. Water Quality Research Journal of Canada, 34(2), 267-280.

Warith, M., Li, X., & Jin, H. (2005). “Bioreactor landfills: state-of-the-art review”. Emirates Journal for Engineering Research, 10(1), 1-14.

Weitz, K. A., Thorneloe, S. A., Nishtala, S. R., Yarkosy, S., and Zannes, M. (2002). "The Impact of Municipal Solid Waste Management on Greenhouse Gas Emissions in the United States." Journal of Air and Waste Management Association, 52 1000-1011.

White J., Ren Q. & Robinson J. (2003). “A framework to contain a spatially distributed model of the degradation of solid waste in landfills”. Waste Management and Research Vol. 21: 330- 345.

White, J., Robinson, J., & Ren, Q. (2004). “Modelling the biochemical degradation of solid waste in landfills”. Waste Management, 24(3), 227-240

Wiles, D. M., Hare, C. W. J., 1997. An alternative daily cover for landfills: a useful tool for increasing drainage efficiency. In Christensen, T. H.; Cossu, R.; Stegmann, R. (eds) SARDINIA 97. Proceedings of the 6th International Landfill Symposium, Lining, Drainage and Stability in Landfills. 355-360.

Waste Management, Inc.(2000) “The Bioreactor landfill the next generation of landfill management.” white paper from WMI -2000

Wreford, K., Atwater, J., and Lavkulich, L. (2000). "The effects of moisture inputs on landfill gas production and composition and leachate characteristics at the Vancouver Landfill Site at Burns Bog." Waste Management and Research, 18(4), 386-392.

Wu, B., Taylor, C. M., Knappe, D.R.U., Nanny, M.A., Barlaz, M. A. (2001). "Factors Controlling Alkyl benzene Sorption to Municipal Solid Waste." Environmental Science and Technology, 35, 4569-4576

Yeşiller, N., Hanson, J. L., and Liu, W. L. (2005). "Heat generation in municipal solid waste landfills." J. Geotech. Geoenviron. Eng., 131 1330.

Yuen, S. T. (1999). “Bioreactor landfills promoted by leachate recirculation: a full-scale study”. Laboratory Investigation, 5, 11.

Yuen, S. T. S., Wang, Q. J., Styles, J. R., & McMahon, T. A. (2001).”Water balance comparison between a dry and a wet landfill—a full-scale experiment”. Journal of Hydrology, 251(1), 29-48.

Zacharof, A. I., & Butler, A. P. (2004). “Stochastic modelling of landfill leachate and biogas production incorporating waste heterogeneity, model formulation and uncertainty analysis”. Waste Management, 24(5), 453-462.

Zeiss, C., & Major, W. (1992). “Moisture flow through municipal solid waste: patterns and characteristics”. Journal of Environmental Systems, 22(3), 211-231.

Zekkos, D. P., Bray, J. D., Riemer, M. F., Kavazanjian, E., and Matasovic, N. (2005). “Field investigation, characterization and unit weight of municipal solid waste.” GeoEngineering Rep., Department of Civil and Environmental Engineering, Univ. of Calif., Berkeley, Berkeley, Calif. (http://waste.geoengineer.org) (accessed July 24, 2006).

Zekkos, D., Bray, J. D., Kavazanjian Jr, E., Matasovic, N., Rathje, E. M., Riemer, M. F., & Stokoe, K. H. (2006). “Unit weight of municipal solid waste”. Journal of Geotechnical and Geoenvironmental Engineering, 132(10), 1250-1261.

Zornberg, J. G., Jernigan, B. L., Sanglerat, T. R., & Cooley, B. H. (1999). “Retention of free liquids in landfills undergoing vertical expansion”. Journal of Geotechnical and Geoenvironmental Engineering, 125(7), 583-594.

Biographical Information

Dipak Tiwari received his Bachelor in Civil Engineering degree in 2005 from Tribhuvan University, Pulchowk Campus, Nepal. After graduating he worked in engineering firm for some years. He came in United States in 2008 for graduate studies. He received his Master's degree in civil engineering with emphasis on geotechnical engineering in December 2011 from Southern Illinois University Carbondale. He was admitted to the doctoral program in the Civil Engineering Department at the University of Texas at Arlington in August, 2011. He worked as an Enhanced Graduate Teaching Assistant for various engineering undergraduate courses. He also got an opportunity to work in various landfills and TxDOT projects.

Along with teaching assistant, he started his research on bioreactor landfill under Dr. Sahadat Hossain in civil engineering department. He also awarded Dean Doctoral Fellowship along with Enhanced Graduate Teaching Assistant from 2011 to 2014 during his whole study period. His main research interest includes landfill waste management, gas generation from waste, renewal green energy, geotechnical earthquake engineering, foundation design and analysis, slope stability analyses, Numerical modeling of earth structures.