LEACHATE RECIRCULATION MODELING USING VERTICAL WELLS IN BIOREACTOR LANDFILLS

by

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Abstract

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Municipal solid waste (MSW) is made up of household and commercial waste. Total MSW generated in USA in 2009 was 243 million tons and 54.3% of this waste generation was landfilled (EPA, 2009). MSW buried in landfills contribute a significant amount of gas (Landfill Gas or LFG) to atmosphere. The landfilled MSW is the third largest methane contributor to the atmosphere.

One way to manage the LFG is the concept of bioreactor landfills. A bioreactor landfill is a controlled landfill where liquid and gas conditions are actively managed in order to accelerate or enhance biostabilization of MSW. Biostabilization of MSW is usually enhanced by leachate recirculation through MSW. Leachate recirculation can be implemented using different techniques. Using vertical wells, which is the topic of this study, is one way to recirculate leachate in a landfill.

Vertical wells are more common in retrofit landfills that implementation of horizontal trenches are not possible or economical. The leachate recirculation system can contain a bunch of single wells, or a bunch of well clusters. In this study, the performance of both single and cluster arrangement was studied.

The main objective of this study was to model the leachate recirculation using vertical wells in order to predict the wetted area around the well. HYDRUS-2D was used to create finite element models that simulate the field condition. The area of interest in this study was the Cefe Valenzuela landfill in Corpus Christi, TX. Currently, the landfill is permitted by Texas Commission on Environmental Quality (TCEQ) to operate as Enhanced Leachate Recirculation (ELR) landfill. The preliminary design details of vertical wells have been used in this study. However, an extensive experimental program carried out to characterize the MSW.

The experimental program showed the fresh or partially degraded state of MSW. However, in some locations such as 40 ft of BH-1 and BH-2 the MSW is mostly degraded. Based on the laboratory test results two set of material properties, one representing the fresh or partially degraded state and another representing the mostly degraded state, were developed and assigned as MSW properties.

Numerical simulations were performed using various leachate quantity and anisotropy factors. Longer lateral extent of leachate distribution is achieved when higher leachate quantity is injected or more anisotropic factor is assumed. Also, the lateral extent was higher in the mostly degraded MSW. In fresh or partially degraded MSW, the affected zone profile is oval shape with the well at top; however, for mostly degraded MSW it is more circular with the well at center.

Leachate injection using a well cluster also was simulated to predict the group behavior of vertical wells. Cluster of two wells was found an efficient arrangement that affects a large area; however, cluster of three wells was not efficient because of the short space below the lowest well.

Finally, simulations were validated based on the City of Denton Landfill. Simulations were performed for the City of Denton using same approach for this study.

The simulation results were matching with the available electrical resistivity images of the recirculated area.

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Chapter 1

Introduction

1.1 Background

Municipal solid waste (MSW) is made up of household and commercial waste, including package wrappings, food scraps, grass clippings, computers, and refrigerators. It does not contain industrial, hazardous, or construction waste. Total MSW generated in USA in 2009 was 243 million tons and 54.3% of this waste generation was landfilled, 33.8% was recycled and composted, and 11.9% was converted to energy (EPA, 2009).

MSW buried in landfills contribute a significant amount of greenhouse gas (especially methane) to atmosphere, causing global warming. Landfill gas (LFG) is combined of almost 50 percent methane and 50 percent carbon dioxide (EPA). In addition, it contains small amounts of nitrogen, oxygen, hydrogen, and nonmethane organic compounds (NMOCs). Thus, landfills generate a significant amount of methane

After natural gas system and domestic livestock landfills are the third largest source that contributes methane to atmosphere, accounting for 17 percent of total methane emissions (2009). Moreover, Forster et al. (2007) stated that heat-trapping capacity of methane in the atmosphere is 25 times greater than carbon dioxide (2007). As a result, methane has the highest capacity to contribute to global warming in comparison with other components of LFG.

Although the methane has potential to contribute to global warming, it can be utilized as a source of power generation if it is properly managed. One way to manage

LFG is the concept of bioreactor landfills. A bioreactor landfill is a controlled landfill where liquid and gas conditions are actively managed in order to accelerate or enhance biostabilization of MSW. The bioreactor landfill significantly increases the degradation of organic wastes and process effectiveness over what would otherwise occur with the landfill (SWANA, 2002). In bioreactor landfills, the biodegradation is usually enhanced by adding moisture through leachate recirculation. The greater rate of biodgradation results more gas production, which can be used for electricity generation.

Leachate recirculation can be implemented using different techniques. The techniques include leachate recirculation using horizontal pipes, horizontal trenches, permeable blankets or vertical wells. In this study, leachate recirculation using vertical wells was the main topic of discussion.

1.2 Research Objective

The major objective of the current study is to predict the extent of moisture distribution after leachate injection by modeling the leachate recirculation using vertical wells in a bioreactor landfill. The simulations have been done to model the performance of vertical wells in the Cefe Valenzuela Landfill, Texas. The specific tasks to accomplish the objectives were as follows:

i. Collection of landfilled solid waste from the Cefe Valenzuela Landfill
ii. Determination of the engineering properties of MSW
iii. Developing the conceptual model for simulations
iv. Modeling of leachate recirculation through vertical wells using HYDRUS-2D
v. Representing the predicted extent of moisture distribution versus time, leachate quantity and anisotropy factor

1.3 Thesis Outline

The thesis report consists of five chapters as follows: Introduction (Chapter1), Literature Review (Chapter 2), Laboratory Methodology (Chapter 3), Numerical Modeling Methodology (Chapter 4), Results and Discussion (Chapter 5), Conclusions and Recommendations for future studies (Chapter 6).

Chapter 2 reviews previous studies about MSW properties, leachate recirculation systems, and numerical simulations of leachate recirculation.

Chapter 3 introduces the methodologies that have been used to characterize the MSW properties. These methodologies have been used to characterize the collected waste samples from the study area.

Chapter 4 represents the methodology that has been used to simulate the leachate recirculation system. In this chapter the modeling procedure and assumptions have been stated. The content is mainly connected to HYDRUS features, the computer software that has been used in this study for predicting the leachate flow.

Chapter 5 focuses on the results and discussions. The leachate distribution extent after leachate injection versus time and leachate quantity as the main outcome is represented in this chapter.

The recommendation for future studies (Chapter 6) summarizes the results and outcomes for the present study and recommendations for future work.

Chapter 2

Literature Review

2.1 Introduction

2.1.1 *Landfilled Municipal Solid Waste*

According to the US EPA, Municipal solid waste (MSW) refers to the stream of waste collected through community sanitation services. MSW is defined as trash or garbage which consists of everyday items discarded after use, such as product packaging, grass clippings, furniture, clothing, bottles, food scraps, newspapers, appliances, paint, and batteries originated from homes, schools, hospitals, and businesses.

Landfilled MSW are wastes recovered from boreholes at different depths. The landfilled wastes are subjected to degradation, which is in most of the studies is a function of age and depth.

2.1.2 *Conventional Lanfills*

In this type of landfills, the basic principle is containment and isolation of MSW in order to minimize the negative effects on environment and human. Conventional landfills which are also called “dry tombs” are designed to encapsulate and drain the waste with no active intervention.

2.1.3 *Bioreactor Landfills*

Bioreactor landfills are landfills that enhance the biodegradation of MSW. Therefore, the stabilization of wastes happens in shorter time. Enhanced Leachate Recirculation (ELR) landfills are bioreactor landfills that use leachate recirculation techniques for enhancement of biodegradation.

2.2 Conventional Landfills versus Bioreactor Landfills

Each kind of landfill, whether conventional of bioreactor, has advantages and disadvantages. One problem with conventional landfills is the long time of MSW degradation and stabilization. According to Lee and Jones-Lee, the waste stabilization in dry tomb landfills may take many decades to hundreds of years (1999).

The biodegradation enhancement in bioreactor landfills can be done with changing some characteristic of MSW within a landfill. Many studies have shown that moisture content is the most important parameter that affects the biodegradation rate. Valencia et al. state that leachate, or the accumulated liquid from an MSW mass, recirculation, for example through horizontal trenches or vertical wells, is the most widely used technique to accelerate the biodegradation (2009). According to Kumar, Chiemchaisri, and Mudhoo, a bioreactor landfill is an MSW landfill that accelerates degradation of the organic waste within first 5-10 years after MSW placement and closure (2011). They state by accelerating the rate of methane generation and converting it to the energy, beside economic aspects there will be environmental benefits (2011).

In many countries, especially developed countries, generated methane in landfills is being converted to energy. The EPA explains as of June 2012, there are 594 landfills operational projects that convert methane to energy and generate approximately 1813MW energy. These 594 projects are examples of a sustainable solution in MSW landfills. Also, Dhokhikah and Trihadiningrum state some examples of energy generation plants in the Philippines, India, Thailand, and China that convert MSW methane to energy (2012). For example, in Beijing an MSW bioreactor landfill with the capacity of 2000 tons/day generates 2.5 MW.

Khire and Mukherjee (2006, p. 1233) state " Leachate recirculation offers many environmental and economic benefits to MSW landfills including: (1) reduction in leachate treatment and disposal costs; (2) greater flexibility in leachate management and treatment; (3) faster biodegradation of waste resulting in increased gas production and quicker waste stabilization and settlement; (4) reduction in the risk associated with contamination from spills during off-site transportation, treatment, and disposal of leachate; and (5) potential reduction in the post-closure care period of the landfill."

Barlaz et al. (2010) mention that degradation enhancement in bioreactor landfills leads to greater settlements in shorter time, which reduces the required disposal space, and higher rate of gas generation that makes the gas recovery for beneficial purposes more feasible.

Khire and Mukherjee (2006) also mention the disadvantages of leachate recirculation. Reduction in factor of safety of landfill slopes due to reduction in shear strength, potential leachate leakage from the side of landfill, and greater leachate pressure head on the liner, which makes the risk of ground water contamination higher, are main disadvantages.

2.3 Recirculation Systems

For leachate recirculation some techniques have been developed so far. According to Khire and Haydar (2005) leachate recirculation techniques can be classified as surface and subsurface applications. Based on the advantages and disadvantages of each technique and before the recirculation system implementation, landfill owners are expected to decide which technique works better.

Surface application techniques mainly are direct spraying of leachate on the landfill surface or surface ponding of leachate, as shown in Figure 2.1. The main disadvantages of these applications are odor problem, poor aesthetics, and potential runoff of leachate into storm water management system (Khire & Haydar, 2005).

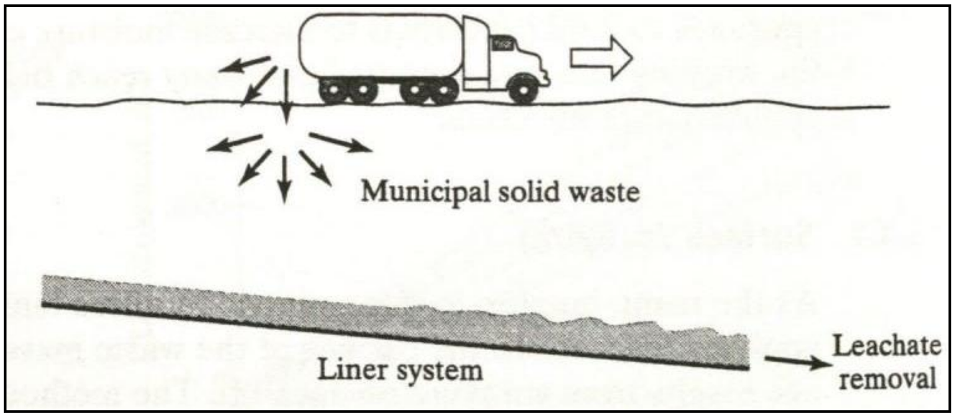


Figure 2.1 Leachate recirculation using surface distribution

Accoring to Khire and Haydar (2005), the main subsurface application techniques are vertical wells, horizontal trenches and permeable blankets. Horizontal trenches are the most used technique in modern lined landfills (Figure 2.2); however, vertical wells are more common in retrofit landfills that implementation of horizontal trenches are not possible or economical.

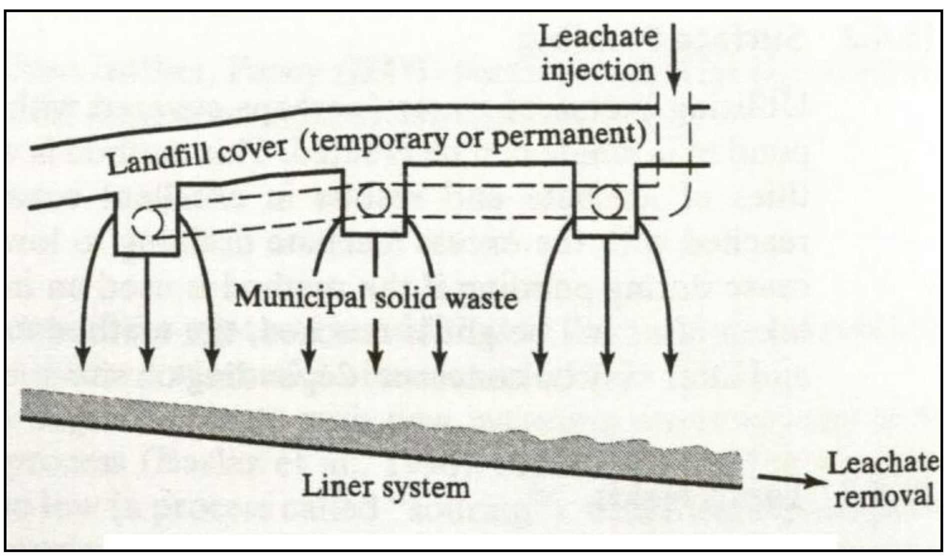


Figure 2.2 Leachate recirculation using horizontal trenches (Qian et al., 2002)

Horizontal trenches, as shown in Figure, are placed below the temporary or final cover. Vertical wells, as shown in Figure 2.3, are wells penetrating the MSW mass. Based on their depth they can be divided to Shallow Wells and Deep Wells. Leachate is pumped and injected through the perforated portion of the well. In the upper lifts, the coverage might be poor; however, it’s probably good at lower depths. The common spacing varies from 10 to 30 meters for shallow wells and 20 to 50 meter for deep wells. As the leachate tends to go downward due to the gravity, deep wells are not as favored as shallow wells (Qian, Koerner, & Gray, 2002).

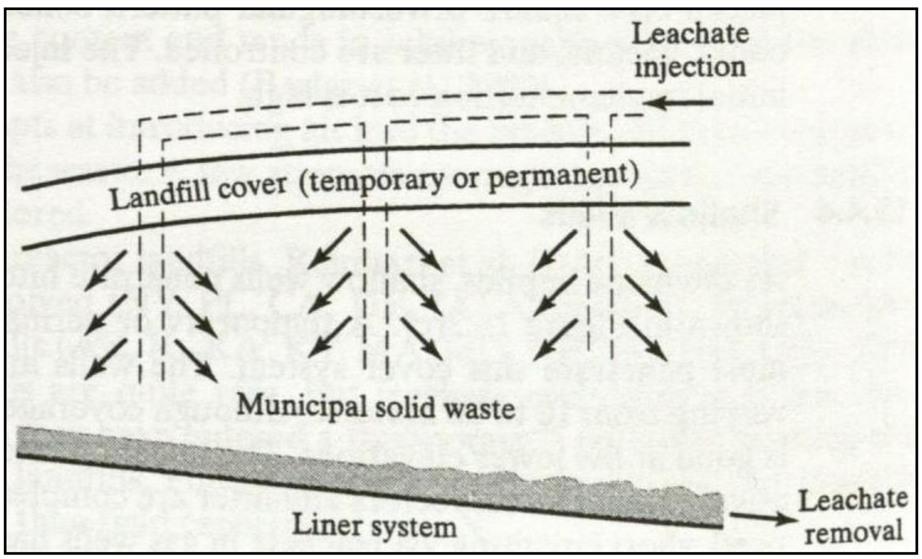


Figure 2.3 Leachate recirculation using vertical wells (Qian et al., 2002)

Neverthless, there is no universally accepted method for leachate recirculation. Comparison of methods based on the existing conditions is the solution. Table 2.1 shows a comparison between the mentioned leachate recirculation methods.

Table 2.1 Leachate recirculation methods (Qian et al., 2002)2002)

Method	Odor/Vector/Liter Control	Injection Rate	Coverage	Injection Cost	Impact on Operations
Surface Spraying	Poor	Fast	Good	Low	Moderate
Surface Ponding	Poor	Fast	Good	Low	Low-to-High
Leach Fields	Good	Moderate	Moderate	Moderate	Low
Shallow Wells	Good	Slow	Poor	High	Low
Deep Wells	Good	Moderate	Moderate	High	Low

2.4 Leachate Recirculation Modeling

Leachate recirculation modeling studies in literature was found rare. The reason might be the complexity in properties of waste materials. Although the purpose of this study is to model the leachate injection using vertical wells, results from the modeling of horizontal trenches and permeable blankets are also worthwhile as both techniques work with leachate flow in a saturated/unsaturated porous media. The first numerical modeling of horizontal trenches and vertical wells was done by McCreanor and Reinhart (1996).

Figure 2.4 shows the processes that affect the leachate movement through a landfill; however, all the processes are not considered in the literature. In all the following studies, these common assumptions and simplifications are made:

i. Richards' equation has been used for predicting the saturated and unsaturated leachate flow in MSW samples.

ii. Waste materials are assumed homogenous. However, waste materials, due to presence of different organic and inorganic materials that are disposed in the landfill, are heterogeneous in hydraulic properties. Moreover, the biodegradation process and the overburden pressure application change the hydraulic properties of the waste.

iii. Waste materials are assumed isotropic, or anisotropy equal to 1. Based on laboratory-scale study conducted by Landva et al. (1998), the anisotropy can reach the upper limit of 10 (K_r / K_z= 10).

iv. The impact of gas production and pressure on fluid flow is neglected.

v. The impact of daily cover soil is neglected. Daily cover soils have a lower hydraulic conductivity. It restricts the vertical spreading of leachate while enhances its lateral spreading.

vi. Channeling effect is neglected. Macropores have much higher hydraulic conductivity than the surrounding micropores because they have relatively low water-entry suctions. Therefore, fluid can flow easier in the macropores (Hardt, 2002). This effect is called channeling or preferential channels as the fluid prefers to enter those channels.

vii. Unsaturated hydraulic properties are assumed as shown in Table 2.2. In all the studies, water retention curve is not measured and the soil-like unsaturated hydraulic properties such as sand properties have been used.

viii. Leachate is assumed as pure water.

ix. Effects of temperature and chemical reactions are ignored.

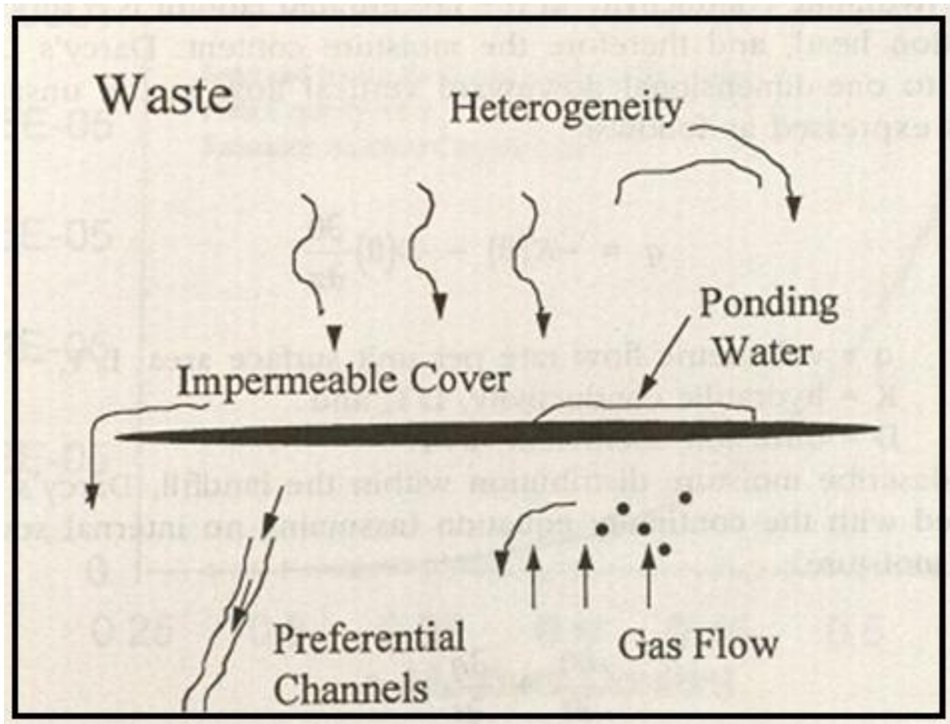


Figure 2.4 processes affecing leachate movement through a landfill (Reinhart & Townsand, 1997)

Table 2.2 Unsaturated Hydraulic Properties of MSW Used in Previous Studies

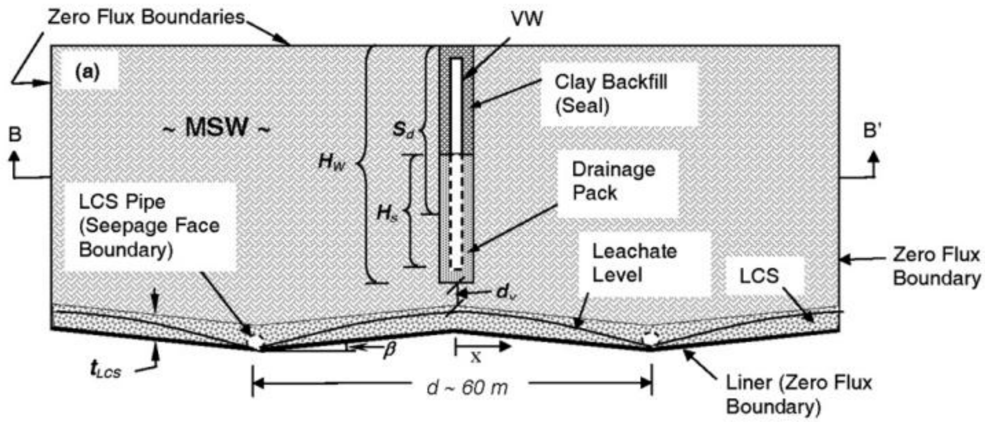
Model	Material	θ_r	θ_s	α (1/m)	n
Haydar and Khire (2005)	Silt loam	0.067	0.45	2	1.41
Khire and Mukherjee (2007)	Silt loam	0.078	0.45	2	1.41
Haydar and Khire (2007)	Silt loam	0.078	0.45	3.6	1.54
Jain (2005)	Sand	0.1	0.5	4	2

2.4.1 *Leachate Recirculation Using Vertical Wells*

For the first time McCreanor and Reinhart modeled the leachate distribution through the wastes using unsaturated conditions (1996). They used SUTRA computer model that uses two dimensional hybrid finite element and integrated finite difference method to simulate the equations of flow. They used the Brook and Corey equations to model the material components of the model.

McCreanor and Reinhart (1996) found that the wetted width was a direct function of the recirculation rate and hydraulic conductivity of the waste. The higher the recirculation rate, the greater the influence distance; however, the recirculation rate may require injection under pressure and may result in leachate seeps. Also, they asserted that vertical wells are inefficient at wetting the upper portion of the landfill. Based on the field data, they stated that leachate infiltration from the well can be increased if the leachate injection is done in on/off dosing cycles.

Khire and Mukherjee (2007) carried out numerical modeling, with HYDRUS-2D, to evaluate the key design variables of leachate recirculation through vertical wells. They evaluated the design parameters based on the wetted width (W_w) of the waste and the pressure head on the liner. According to Khire and Haydar (2005) achieving the greatest W_w is important because it significantly affects the operational costs . Figure 2.6 shows how they determined Ww based on the the isoline for 90% degree of saturation. Also , they assessed the parameters based on the results under steady-state flow conditions. They studied (1) hydraulic conductivity of the waste and vertical well backfill, (2) liquid injection rate and dosing frequency, (3) well diameter, screen height, and screen depth, and (4) hydraulic conductivity and slope of the leachate collection system and spacing of the leachate collection pipes.



Notes:
1. VW = Vertical Well; LCS = Leachate Collection System; and MSW = Municipal Solid Waste.
2. S_d = Screen depth (measured from the surface to the centre of the well screen)
3. H_s = Screen height (screened portion of the well is the constant flux boundary)
4. H_W = Total height of the well
5. d_v = Vertical clearance between the bottom of the well encasing and the top of the LCS
6. NOT TO SCALE.

Figure 2.5 Conceptual model for numerical simulation of leachate recirculation in MSW landfill using a vertical well (Khire & Mukherjee, 2007)

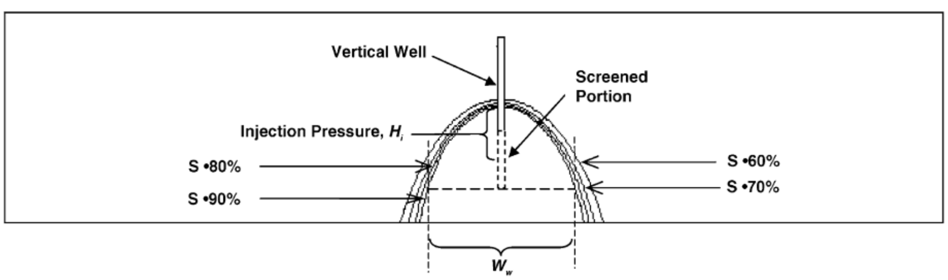


Figure 2.6 Determination of W_w (Khire & Mukherjee, 2007)

Khire and Mukherjee (2007) developed a conceptual model for their numerical simulation, as shown in Figure 2.5. They assumed waste as homogenous and isotropic porous medium. They used both saturated and unsaturated parameters to model the materials. They used silt loam unsaturated hydraulic conductivity parameters for the MSW because it has the closest saturated hydraulic conductivity to saturated hydraulic conductive of MSW. They stated that Van Genuchten fitting parameters, Θ_r , s, 1/m, and n, do not have a significant influence in W_w or pressure on the liner under steady-state flow conditions; however, they are influential parameters under transient flow conditions.

According to Khire and Mukherjee (2007) steady-state flow condition is the moment that the injected leachate flux equates the total leachate flux seeping from the leachate collection system. They mention that although steady-state flow condition can be rarely achieved in the field, its results are more trustworthy when the reliable unsaturated hydraulic properties of the waste do not exist.

Finally Khire and Mukherjee determined the influence of different parameter on W_w and the pressure head on the liner. They used leachate injection rates ranged from 5.5 m^3/d to 55 m^3/d to accommodate typical and high liquid injection rates. They asserted that W_w is primarily a function of liquid injection rate, hydraulic conductivity of waste and on/off frequency used for liquid injection. They concluded that the greater hydraulic conductivity of waste results the lower W_w, as shown in Figure 2.7. Also, they stated that the greater liquid injection rate and on/off frequency result the greater $W_{w.}$ Well diameter, height of the screened portion of the well and depth of the well were found neutral. Also they concluded that the maximum pressure head on the liner is a function of the liquid injection rate, hydraulic conductivity of the leachate collection system and waste, slope of the leachate collection system, horizontal distance between leachate collection pipes and vertical distance between the bottom of the well and top of leachate collection system. They found the hydraulic conductivity of the leachate collection system as the most important parameter on the pressure head on the liner.

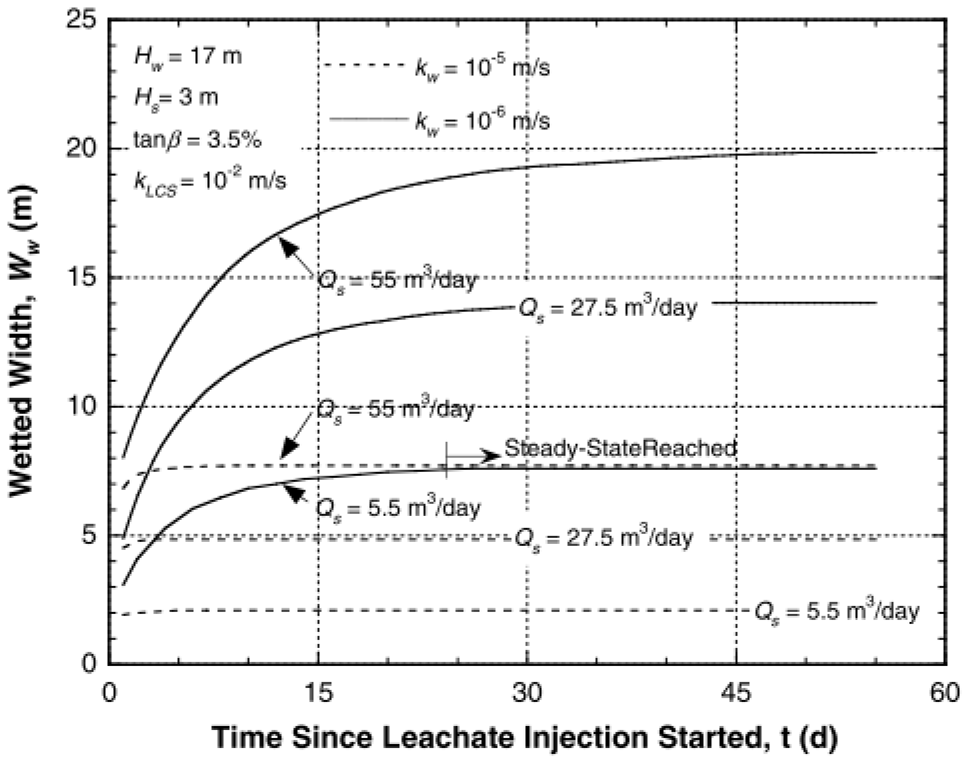


Figure 2.7 Simulated Wetted Width as a Function of Leachte Injection Flux (Khire & Mukherjee, 2007)

In another study, Jain developed a mathematical modeling of moisture addition at a bioreactor landfill using vertical wells (2005). The conceptual model he used is represented in Figure 2.8.He used the saturated-unsaturated flow and transport model (SUTRA) to simulate the saturation profiles around a vertical well after leachate injection. The purpose of his study was to assess the impact of different parameters on moisture and pressure distribution around a vertical well.

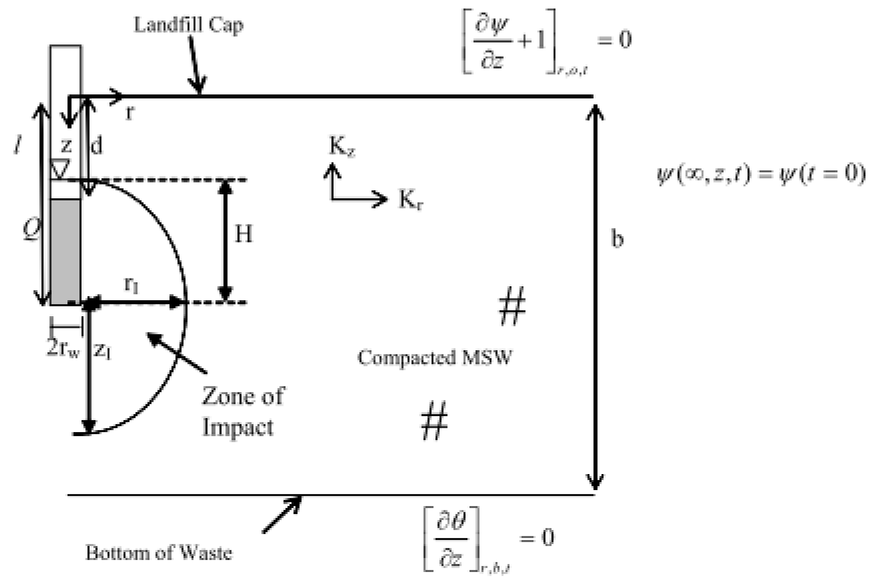


Figure 2.8 Conceptual model for numerical simulation of leachate recirculation in MSW landfill using a vertical well (Jain, 2005)

Jain (2005) formulated some dimensionless parameters to reduce the number of variables. Table 2.3 shows the definition of dimensionless parameters he used. He investigated the impacts of flow rate, waste hydraulic conductivity, anisotropy ratio, well dimensions (screen length and radius), and duration of moisture addition.

Jain used unsaturated flow properties typical of sand for MSW. To evaluate the impact of the unsaturated flow properties, he compared using of clay typical properties (α=0.15 m^{-1} and n=1.5) and sand typical properties (α=0.4 m^{-1} and n=2) . He concluded that the extent of lateral movement in clay was 10 % greater than that of sandy soil; however, the saturated area was bigger for the sandy soil.

Table 2.3 Dimensionless parameters used by Jain (2005)

Parameter	Formulation
Anisotropy Ratio	$a = \frac{K_r}{K_z}$
Dimensionless time	$\tau = \frac{K_z * t}{S_s * r_w^2}$
Dimensionless radial flux	$q_r = \frac{Q}{r_w * (l-d)K_r}$
Note : S_s= Specific storage, r_w= Well radius, Q= Flow rate, *l*-*d*= Screen length	

Finally, he concluded that the pressure at the bottom of the well is dependent on q_r. Also he concluded that the pressure in the well was relatively insensitive to τ. He found that the lateral and vertical extents of moisture distribution is sensitive to q_r and τ. Also, the lateral extent is sensitive to a. Jain (2005) stated that unsaturated flow properties, or van Genuchtens' parameters, have a minor impact on the parameters of interest of his study. He cited that for a conservative design, use of lower hydraulic conductivity, lower anisotropy ratio, and unsaturated media properties typical of sandy soil might be used when there is no estimation of these parameters.

Also, Jain (2005) studied a full scale bioreactor landfill in Florida with existence of 11 injection wells having different heights. He found that higher leachate flow rates could be achieved through shallow wells compared to deep wells. He explained that the lower rate for deeper wells can be due to the lower hydraulic conductivity of the waste due to higher overburden pressure.

2.4.2 *Leachate Recirculation Using Horizontal Trenches*

Haydar and Khire (2005) modeled leachate recirculation using horizontal trenches. Figure 2.9 shows their conceptual model they developed using HYDRUS-2D.

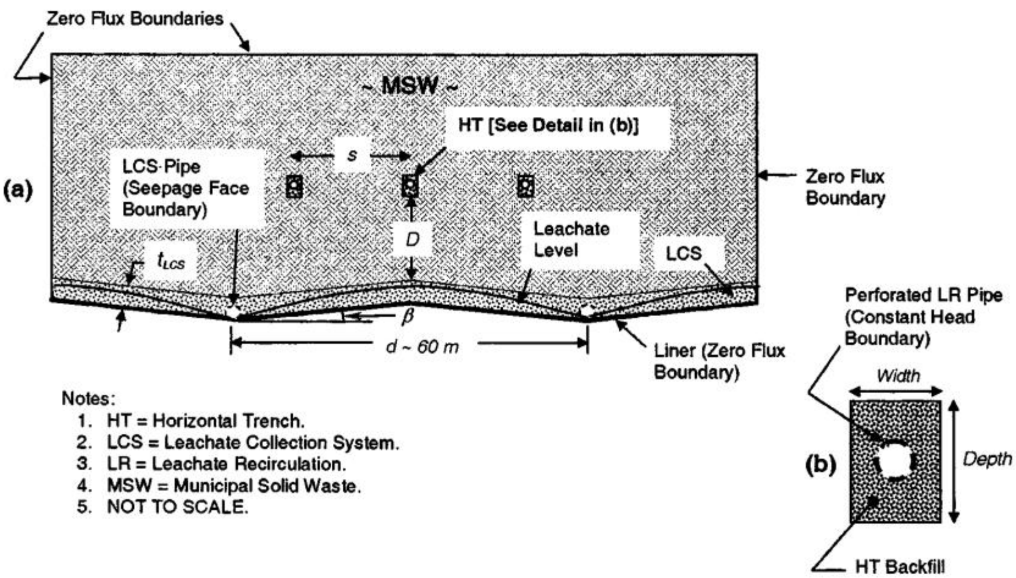


Figure 2.9 Conceptual model used for numerical simulation of leachate recirculation in waste landfill using a horizontal trenches (Haydar and Khire, 2005)

Haydar and Khire (2005) concluded that Logarithm of leachate flux and leachate injection pressure head have a curvilinear relationship and leachate flux is directly proportional to the hydraulic conductivity of MSW condition. Also, they stated that if the hydraulic conductivity of trench backfill is equal or greater than that of MSW, it does not affect the leachate flux.

Haydar and Khire (2005) validated their results with previous studies. They compared the results with McCreanor (1998) and Bachus et al. (2002) results that used SUTRA-2D and VS2DI to develop the same model. They found their results consistent with the previous studies.

Manzur simulated the leachate injection using a horizontal trench with HYDRUS (2006). Also, he compared the numerical modeling results with the field measurements using electrical resistivity imaging. The landfill location he studied was placed in Texas.

Manzur studied the effect of MSW saturated hydraulic conductivity and the injected flux on the wetted width (2013). He assumed the injected flux value based on the field practice. Also, he conducted the resistivity imaging at the field one day after the leachate injection so he was able to compare them. Figure shows the comparison between the numerical modeling results and the field data. He found the wetted width in the field higher than the simulated wetted width, as shown in Table 2.4.

Table 2.4 Comparison of the actual and simulated wetted width (Manzur, 2013)

Wetted Width (ft)		
Field data		Simulation
Left	Right	Both sides
85	75	60

Manzur (2013) concluded that distribution of flow was not uniform across the pipe section based on the resistivity imaging results. Also he cited that the heterogeneity and anisotropy of the waste material plays significant role in leachate distribution. Due to the presence of heterogeneous waste materials, moisture has a tendency to follow preferential channels. Therefore, the extent of recirculation can vary at different locations of the landfill based on the type of waste, compaction level. Finally, he stated that the distribution of the simulated flow is more uniform and cover a narrow area compared to the actual field condition.

2.4.3 *Leachate Recirculation Using Permeable Blankets*

Haydar and Khire (2007) modeled leachate recirculation using permeable blankets. They also validated the results with the field data from a landfill in Michigan.

They used HYDRUS-2D to simulate the travel and pressure head of injected leachate in permeable blankets. Figure 2.10 shows their conceptual model used for numerical simulation. They assumed a range of values for unsaturated hydraulic conductivity because they did not measure the water retention curve for the waste materials.

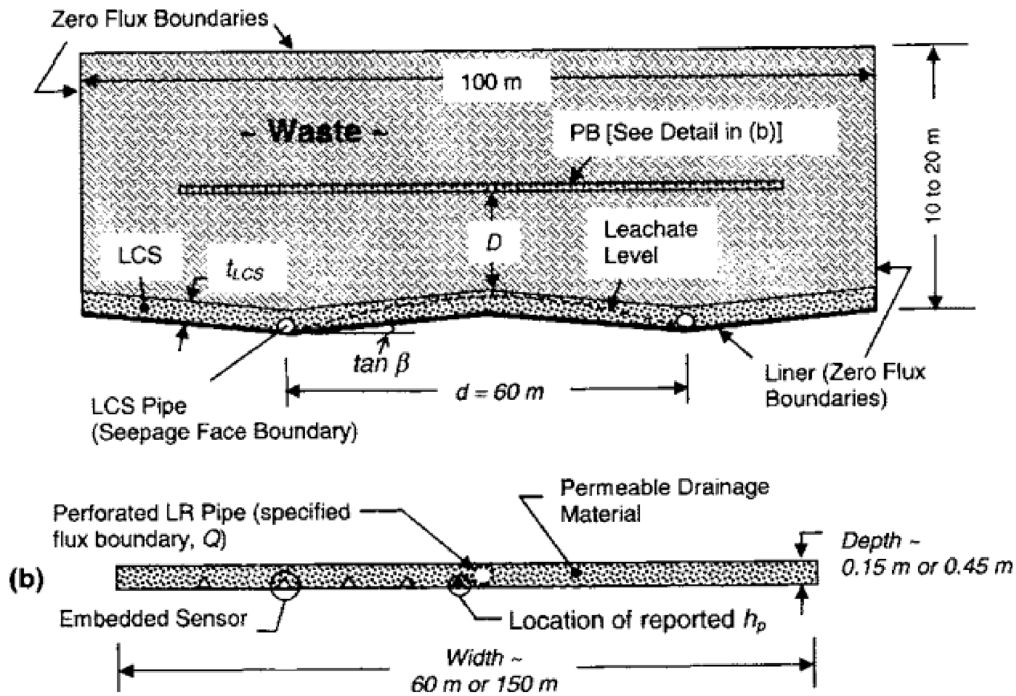


Figure 2.10 Conceptual model used for numerical simulation of leachate recirculation in waste landfill using a horizontal permeable blanket (Haydar and Khire, 2007)

Although the accurate simulation was not possible because the hydraulic conductivity of the waste materials were unknown, the predicted pressure heads using the numerical model were consistent with the field data for the assumed input.

2.5 Characterization of Landfilled Municipal Solid Waste

Analysis and design of MSW landfills is significantly dependant on the engineering properties of waste materials. Therefore, the characterization of MSW is a critical step before doing any modeling; however, determining the correct values of MSW engineering properties is not easy. According to Fassett et al. (1994) the following conditions of MSW makes determination of MSW engineering properties difficult:

i. MSW properties are widely variable due to its inconsistent and heterogeneous composition.
ii. It is very difficult to obtain a sample that is big enough to represent the field condition.
iii. The erratic nature of the waste particles makes sampling and testing difficult.
iv. Waste properties change with time, depth, and location

Nevertheless, the laboratory test results are compared with the values offered in literatures to check whether they are in a reasonable range.

2.5.1 *Physical Composition*

Physical composition of MSW represents the MSW constituents and their weight percentages in an MSW mass. Physical composition is one of the most important parameters in the landfill behaviors, such as hydraulic conductivity or biodegradation behaviors.

For classifying an MSW mass into its constitutive components, MSW classification is required. Although different classifications have been used in different studies, here two classification systems are represented. Landva and Clark (1990) developed the classification based on their observation that some waste constituents are readily biodegradable, some are slowly biodegradable, and some are not biodegradable or very slowl degradable. Figure 2.11 shows the MSW classification based on the biodegradability. In Table 2.5, the Geosyntec (1996) classification is shown which classifies the MSW constituents based on several criteria.

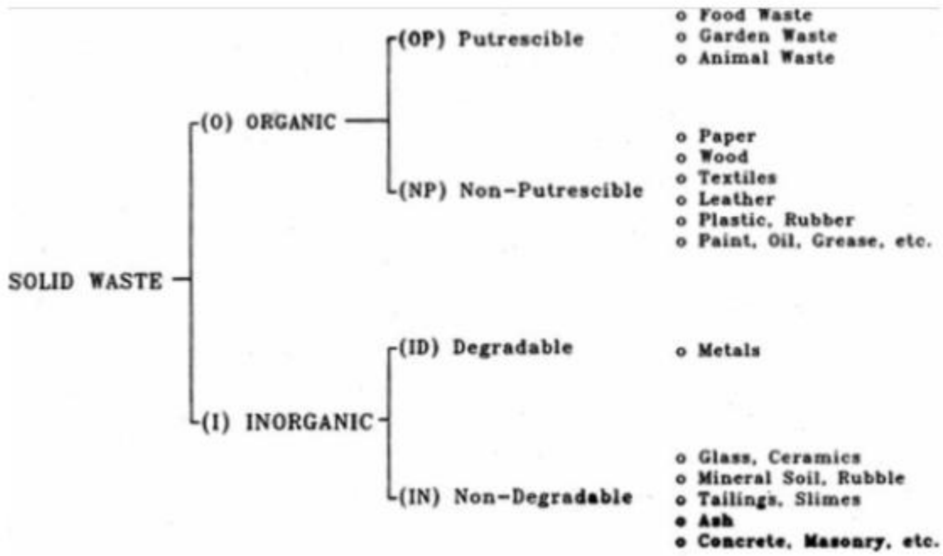


Figure 2.11 Solid Waste Classification Based on their Biodegradability (Landva and Clark, 1990)

Table 2.5 Landfill Field Waste Classification Scheme (Geosyntec,1996)

Moisture Content		Composition	
1	Dry dump moisture level	1	Household-paper and plastics
2	Wet moisture levels	2	Putrescible organics
3	Standing water	3	Concrete, bricks
		4	Wiring
Compaction		5	Metal
1	Slight-refuse easily falls out of bucket auger	6	Nonferrous Metal
		7	Tiers
2	Moderate-refuse falls out of bucket ager upon impact	8	Asphalt
		9	Soil
3	Heavy-refuse falls out of bucket auger only after being struck multiple times	10	Medical
		11	Indistinguishable
		12	Glass
		13	Other (specify)
Degradation			
1	None-newspaper very legible, no refuse discoloration		
		Structure	
2	Slight-some newspaper still legible, discoloration	1	Layered
		2	Encapsulated
3	Moderate-newspaper partly legible, highly discolored	3	Fibrous
		4	Interlocked
4	High-newspaper highly faded gray to black	5	Indistinguishable

Physical composition in different places can be different. Figure 2.12 shows the waste stream in different countries based on their development level. One fact that is concluded from the figure is the high content of biodegradable constituents, specifically organics or food wastes, in middle-income countries .According to Khatib, the reason for difference in percentage of organic materials is different lifestyle in different countries (2011, p. 39). He states that in high- income countries, processed and homemade foods are less used (2011, p. 39). Consequently, the percentage of biodegradable constituents is less than middle-income or developing countries where people use more homemade processed foods.

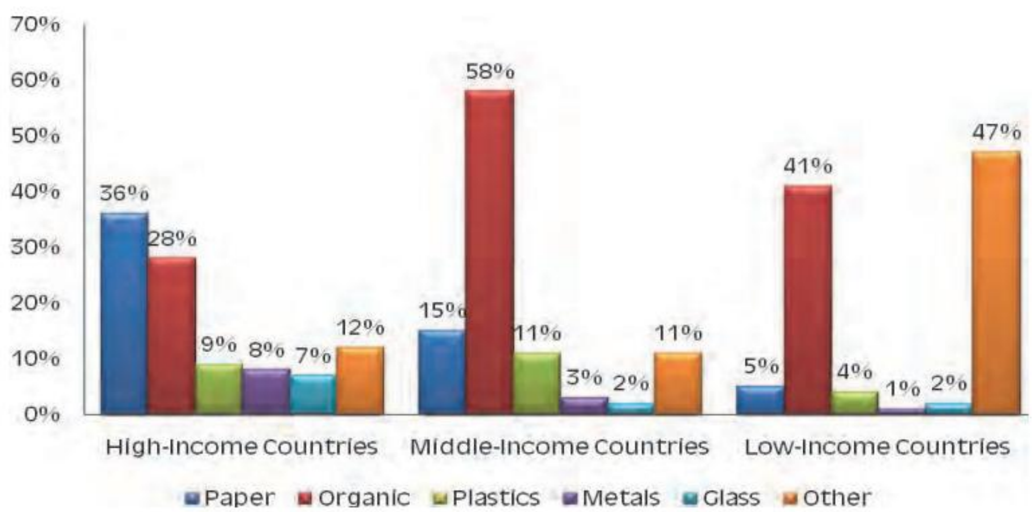


Figure 2.12 MSW Composition Depending on Income (United Nations Department of Economic and Social Affairs, 2010)

Waste composition, due to its importance in landfill behavior, has been studied in different countries. Staley and Barlaz (2009) studied composition of municipal solid waste in the United States. They researched waste composition of eleven states in the United

States to find if the composition is different in different states or not. As a result of waste composition comparison, they asserted although there were some minor differences in waste composition in different states, the decomposable organic content was almost same for all the states. Figure 2.13 shows the waste stream in the USA based on the EPA report (2011). In another study, Dhokhikah and Trihadiningrum studied solid waste management in Asian developing countries (2012). The study shows high content of degradable wastes in the waste streams in 11 cities, as shown in Table 2.6. Food wastes constitutes %15 of the total waste in the USA; however, the average of decomposable organics for the 11 mentioned cities is %58. Consequently, both studies confirm dependency of the waste stream to income, social and economical development level. Finally Table 2.7 summarizes the waste stream in the US and Texas.

Table 2.6 MSW Decomposable Organic Percentage in Asian Developing Cities

City (Country)	Decompostable Organic %
Surabaya (Indonesia)	72
Jakarta (Indonesia)	68
Allahbad (India)	45
Puducherry (India)	42
Kathmanda (Nepal)	71
Bangok (Thailand)	42
Phuket (Thanland)	49
Yala (Thailand)	49
K. Lumpur (Malaysia)	61
Rasht (Iran)	80
Dhaka (Bangladehs)	68

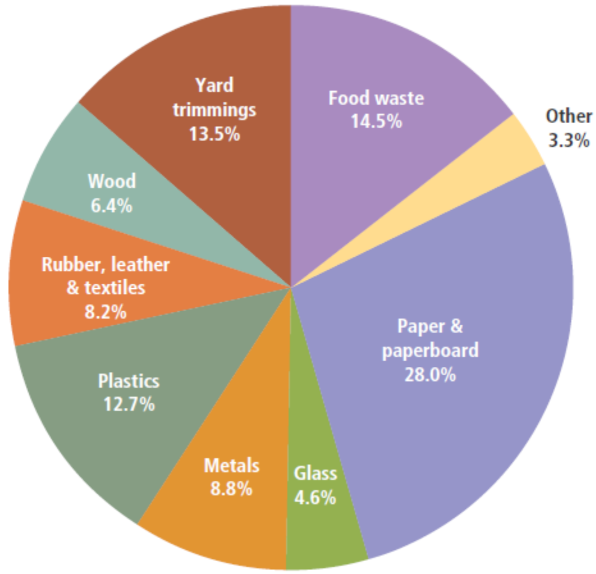


Figure 2.13 Total MSW Generation in the USA in 2011 (EPA, 2011)

Table 2.7 USA and Texas MSW Stream

Constituent	USA	Texas
Paper	28	36
Plastic	12.7	8
Yard Trimmings	13.5	20
Metal	8.8	5
Food	14.5	9
Wood	6.4	6
Glass	4.6	5
Other	11	11

2.5.2 *Moisture Content*

Moisture content in a landfill depends on parameters such as precipitation, type of capping, waste type, site management, and the geological and hydrogeological conditions of the site (Yochim et al., 2013). Moisture content of wastes in a landfill varies both spatially and temporally.

Moisture content in conventional dry tomb landfills is between %15 - % 40 (Tchobanoglous et al, 1993); however, optimum moisture content for biodegradation of decomposable wastes is above 65%, w/w (Tchobanoglous et al., 1993; Rodriguez et al., 2001; and Imhoff et al., 2007).

Biodegradation of MSW in a landfill is highly dependent on its moisture content. Barlaz states “high moisture will promote the dissolution and mixing of soluble substrates and nutrients and will also provide a mechanism for microbial transport within a landfill” (1997, p. 543). Indeed, moisture increases the possibility of contacts between decomposable organic materials and microorganisms that digest them.

During anaerobic decomposition of MSW, moisture is being consumed by microorganisms and is being transformed to some productions like methane. Consequently, moisture content and decomposable organics are expected to be greater in fresh MSW. Also, in the MSW stream with higher readily biodegradable organics like foods, the moister content is higher. As a result, greater moisture content is expected for the waste stream in developing countries. Carboo and Fobil (2005) studied the moisture content in three zones in Accra Metropolitan, Ghana. They found the moisture content of %62.2, %46.9, and %39.8; however, the average moisture content in North America is generally %26.

Based on Samir (2011), average wet weight basis moisture content was %25 for 9-25 year landfilled samples in City of Denton Landfill, in Texas.

Raga and Cossu (2013) determined the moisture content of three landfilled samples at depth of 5 m, 10 m, 15 m, corresponding to approximately 5, 10 and 15 years of age of the landfill . The moisture contents were respectively %31, %27, and %28.

Table 2.8 Moisture Content of Landfilled Samples (Wu et al., 2012)

Borehole	Depth (m)	Age (yr)	w/w MC (%)
A	6-7	3	36.6
	10-11	4	36.1
	14-15	5	37.6
	18-19	6	35
	22-23	8	27.3
B	6-7	3	33.6
	10-11	4	29.3
	14-15	5	34.2
	18-19	6	42.7
	22-23	8	28.8
C	6-7	3	31.3
	10-11	4	36.6
	14-15	5	31.1
	18-19	6	36.8
	22-23	8	24.7

Wu et al. (2012) determined moisture content of landfilled samples from a landfill located south of Beijing, China. Wet Weight (w/w) basis moisture contents of landfilled samples are listed in Table 2.8 with their corresponding age and depth.

Hogland et al. (2004) measured the moisture content of landfilled samples in Måsalycke landfill, in Sweden, at three different depth. He stated the moisture content of 17-22 year old samples were almost same. Moisture contents are reported %29, %29, and %30 (w/w) at 0.6, 3, and 7m depth, respectively.

Quaghebeur et al. (2013) studied moisture content of 14-29 year old landfilled samples from 8 m to 13 m with. The average moisture content of the waste samples at one location varied between 48 and 66%, w/w. They observed some dry layers close to the saturated layers. They stated that the moisture content significantly changes in different areas because of poor-drainage or impervious layers in the landfill.

2.5.3 *Unit Weight*

MSW unit weight is one the most important parameters to analyze landfill systems. According to Zekkos et al. (2006) although literatures present scattered unit weight values, with a consistent waste composition, waste handling practice, and predictable confining stress effects a relation between unit weight and depth can be found in many landfills. Zokkos states that waste composition and landfill operational practice such as compaction effort, cover soil placement, liquids management during waste placement are some parameters that affect the unit weight. Also confining stress which is represented by a depth term and degradation are another parameters affecting the unit weight.

Landva and Clark (1990) conducted unit weight tests in different landfills in Canada and stated the in-situ unit weight range is between 6.8 to 16.2 kN/m^3. They also cited that the possibility of weighing error is high. Therefore, based on the error, they determined the unit weight range between 7 to 14 kN/m^3.

Bulk unit weight of MSW in different locations and conditions are listed in Table 2.9. It shows the wide range of unit weights due to different conditions. A degraded waste mass has a higher unit weight because it includes more fine particles as a production of biodegradation (Hossain et al., 2008). Table 2.10 shows the unit weight in different degradation phases.

Table 2.9 Bulk Unit Weight in Different Countries (Dixon & Jones, 2004)

Country	Measured bulk unit weights (kN/m^3)	Comments	References
United Kingdom	6	• Compacted in 2 m lifts using steel wheeled 21 tonne compactor	Watts and Charles (1990)
	8	• 0.6 m lifts using same compactor as above	
Belgium	5–10	• Common compaction practice	Manassero et al. (1996)
France	7	• Upper layers of fresh (non-degraded) MSW	Gourc et al. (2001)
USA	6–7 14–20	• Fresh MSW after initial placement • Degraded waste with high % of soil like material	Kavazanjian (2001)

Table 2.10 Unit Weight in Different Degradation Phase (Hossain et al. 2008)

Phase	Unit Weight (kN/m3)
I	8.5-9.1
II	9.2-9.8
III	10.1-10.3
IV	10.7-11.2

2.5.4 *Hydraulic Conductivity*

The most important factor affecting the Hydraulic conductivity of MSW is its void ratio or interconnected void spaces. According to Reddy (2009) as the degraded MSW includes the finer particles, the hydraulic conductivity of landfilled MSW is lower than fresh MSW. Moreover, as MSW is heterogeneous and degradable that makes the hydraulic conductivity spatially and temporally more variable. Particle sizes, material type, degree of saturation are other parameters that influence the permeability (Hossain et al., 2008). Also horizontal hydraulic conductivity can be greater than the vertical, because of horizontal stratification of the MSW mass due to compaction of thin lifts at the field.

Hossain et al. (2008) conducted hydraulic conductivity tests with constant head on 4 lab scale bioreactor landfill to observe the variation of hydraulic conductivity with degradation. As it is shown in Table 2.11 the hydraulic conductivity decreased from 0.0088 cm/s to 0.0013 cm/s as degradation advanced.

Table 2.11 Variation of Permeability with Degradation (Hossain et al., 2008)

Phase	Coefficient of Permeability (cm/s)
I	8.80E-03
II	7.30E-03
III	2.50E-03
IV	1.30E-03

Wu et al. studied unsaturated hydraulic conductivity of MSW ant its variation with depth and age (2012). In Figure 2.14, the shallow layer is 1-4 m depth and 3 years old, the middle layer is 11-14 m and 6 years old, and the deep layer is 22-25 m and 10 years old. As the effective moisture content decreases, the relative hydraulic conductivity decreases. Also, they stated that as landfill age increases, the waste becomes more homogenous and it shows more silt-loam like properties.

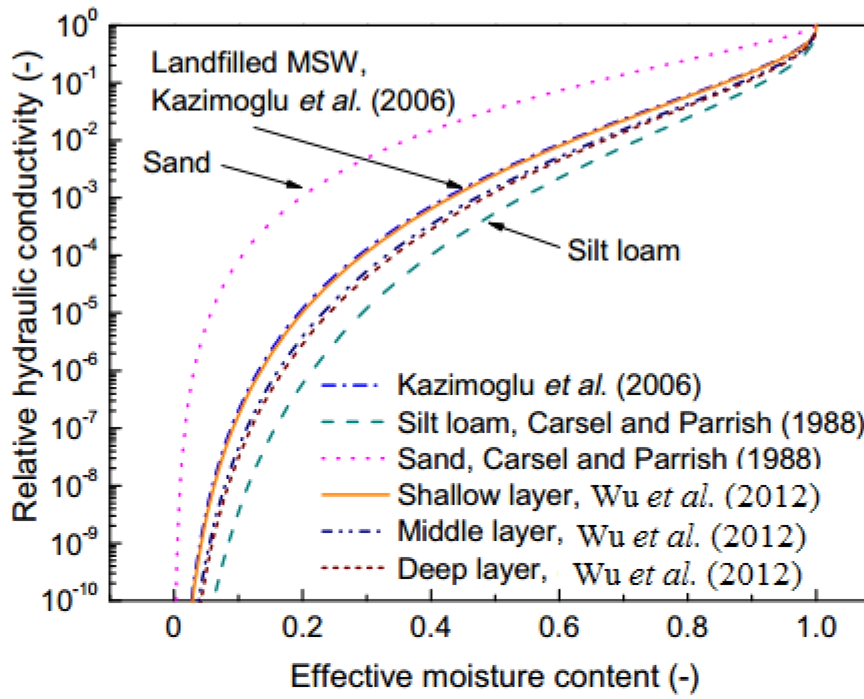


Figure 2.14 Relative hydraulic conductivity values in different studies (Wu et al., 2012)

Jain et al. (2006) measured hydraulic conductivity in 23 locations in a landfill in Florida, using the borehole permeameter test, and found the saturated hydraulic conductivity of MSW between $5.4 \times 10^{-6}\ to\ 6.1 \times 10^{-5}\ cm/s$. As it is shown in Table 2.12, a broad range of hydraulic conductivity values are mentioned in previous studies that is because the extremely variable and heterogeneous nature of MSW.

Table 2.12 Hydraulic Conductivity of MSW in Literatures (Jain et.al, 2006)

Measurements	Hydraulic conductivity (cm/s)	Test
Laboratory		
Fungaroli and Steiner (1979)	10^{-4}–10^{-2}	Constant head
Korfiatis et al. (1984)	8×10^{-3}–1.3×10^{-2}	Constant head
Noble and Arnold (1991)	8.4×10^{-5}–6.6×10^{-4}	Constant head
Bleiker et al. (1993)	1×10^{-8}–3×10^{-7}	Falling head
Chen and Chynoweth (1995)	4.7×10^{-5}–9.6×10^{-2}	Constant head
Landva et al. (1998)	2×10^{-6}–2×10^{-3}	Constant head[a]
	4×10^{-5}–1×10^{-3}	Constant head[b]
Powrie and Beaven (1999)	3.7×10^{-6}–1.5×10^{-2}	Constant head
Jang et al. (2002)	2.91×10^{-4}–2.95×10^{-3}	Constant head
Field		
Ettala (1987)	5.9×10^{-3}–0.25	Pumping test (Jacob method)
Oweis et al. (1990)	1.0×10^{-3}–2.5×10^{-3}	Pumping test (Theis method)
Shank (1993)	6.7×10^{-5}–9.8×10^{-4}	Slug test
Townsend et al. (1995)	3×10^{-6}–4×10^{-6}	Zaslavasky wetting front
Landva et al. (1998)	10^{-3}–3.9×10^{-2}	Flow nets
Wysocki et al. (2003)	1.2×10^{-5}–6.3×10^{-4}	Pumping test

[a]Hydraulic conductivity in the vertical direction.

[b]Hydraulic conductivity in the horizontal direction.

Table 2.13 Hydraulic Conductivity of MSW in Literatures (Stoltz et.al, 2010)

Source Measurement method	Tested MSW	Dry density ρ_d [Mg/m^3]	Compression σ' [kPa]	Hydraulic conductivity K_w [m/s]
Chen and Chynoweth (1995)	Reconstituted US MSW	0.16–0.48	–	9.6×10^{-4}–4.7×10^{-7}
Powrie and Beaven (1999)	Unshredded MSW from tipping face of a UK landfill	0.39 0.72	40 600	1.5×10^{-6}–3.4×10^{-5} 2.7×10^{-7}–3.7×10^{-8}
Jang et al. (2002)	MSW from a Korean landfill	0.80–1.20	–	3.0×10^{-5}–2.9×10^{-6}
Durmusoglu et al. (2006)	Type 1 MSW samples from a US landfill in Brazos Country, Texas	–	123–369	1.2×10^{-4}–4.7×10^{-6}
Reddy et al. (2009)	Samples drilled from a US landfill in Davis Junction, Illinois	0.41–1.33	0–276	2×10^{-3}–4.9×10^{-7}
Stoltz et al. (2010)	Fresh MSW from the same landfill French fresh MSW	0.32–0.96 0.36–0.60	0–276 0–200	2×10^{-3}–7.8×10^{-7} 1.6×10^{-3}–4.9×10^{-6}
		0.49–0.60	100–200	1.0×10^{-4}–1.1×10^{-5}

2.5.5 *Volatile Solids*

The weight loss after ignition of dry MSW in a certain temperature is called Volatile solids (VS). According to Mehta et.al (2002) VS includes both degradable, such as cellulose, hemicelluloses, and recalcitrant organic compounds, such as lignin, plastic, rubber, etc. This test can be used to determine the degradation state of MSW and its gas potential.

Kelly et al. (2006) conducted an experimental program to find the best parameters to determine the bio-stability of landfill MSW. They collected 12 samples from different landfills in the US with different ages, from fresh to 11 years old samples. Cellulose, lignin, BMP, VS along with plastics, and the cellulose to lignin ratio were parameters that were compared. According to Kelly et al. (2006) VS test is an easy, fast, and economical test to determine the stability of MSW in a landfill in comparison with Biochemical Methane Potential (BMP) test which is complex and time consuming. Kelly et al. mention that VS seems the best parameter to determine the waste bio-stability. They state although BMP results are good indicator of degradation state, they are affected by the variability of inoculums type.

According to Kelly et al. (2006) the lower VS, means the more stable waste because waste samples with low volatile solids contain little organic material. Therefore, they observed lower values of VS, cellulose, lignin and BMP for bioreactor landfills that wastes degrade faster within them. Also it is expected that older wastes have lower VS values as it is shown in Figure 2.15.

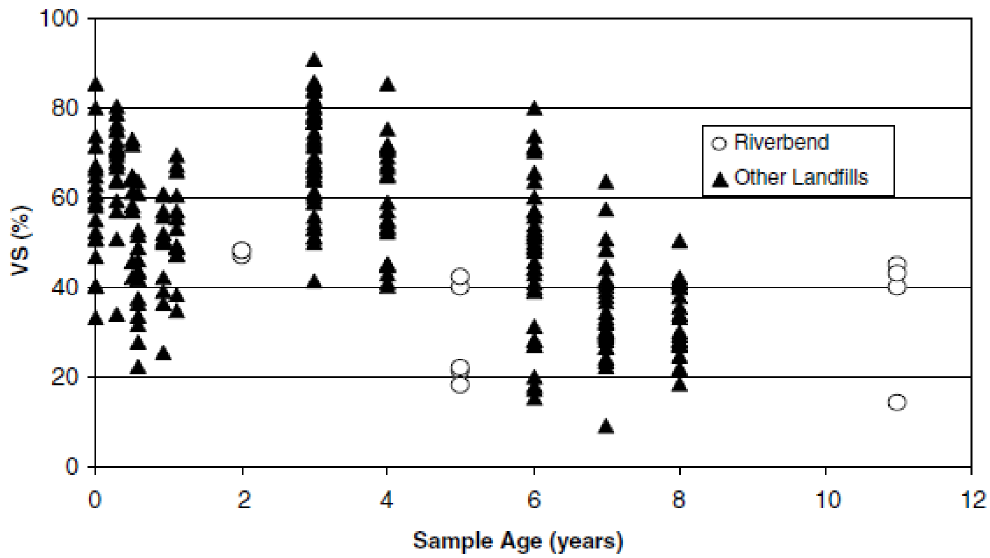


Figure 2.15 VS Values versus Age

2.5.6 *Unsaturated Hydraulic Conductivity of Municipal Solid Waste*

In this study, MSW is the porous media. Consequently, hydraulic properties of MSW are of paramount importance. The van Genuchten-Mualem function has been used to predict the unsaturated hydraulic conductivity of MSW while saturated hydraulic conductivity of MSW is measured based on the laboratory tests. The van Genuchten-Mualem function needs the van Genuchten fitting parameters to calculate the unsaturated hydraulic conductivity.

The van Genuchten fitting parameters are obtained from the soil-water characteristic curve (SWCC). SWCC represents the relationship between soil suction and water content for a soil sample. There are mathematical models that represent the SWCC. These mathematical models use fixed points pertaining to water content or suction at specific condition, such as saturation, and two or more empirical or semiempirical fitting constants that can simulate the general shape of the curve between the fixed points. The van Genuchten model fitting parameters are named n,m, and α. Equation 4.2 represents the the van-Genuchten mathematical model of SWCC for the simulated materials (van Genuchten, 1980).

$$\theta = \theta_r + \frac{\theta_s - \theta_r}{(1 + |\alpha\psi|^n)^m} \tag{4.2}$$

Where:

$\boldsymbol{\theta_r}$= Residual volumetric water content [dimensionless]

$\boldsymbol{\theta_s}$= Saturated suction head [dimensionless]

Ψ =Metric Suction head [L]

α [1/L] , n, m = Fitting parameters

Same methodology can be used to estimate the unsaturated hydraulic conductivity of the MSW samples. MSW samples can be used instead of soil samples to develop the Water retention curve (WRC), instead of SWCC, as shown in Figure 2.16.

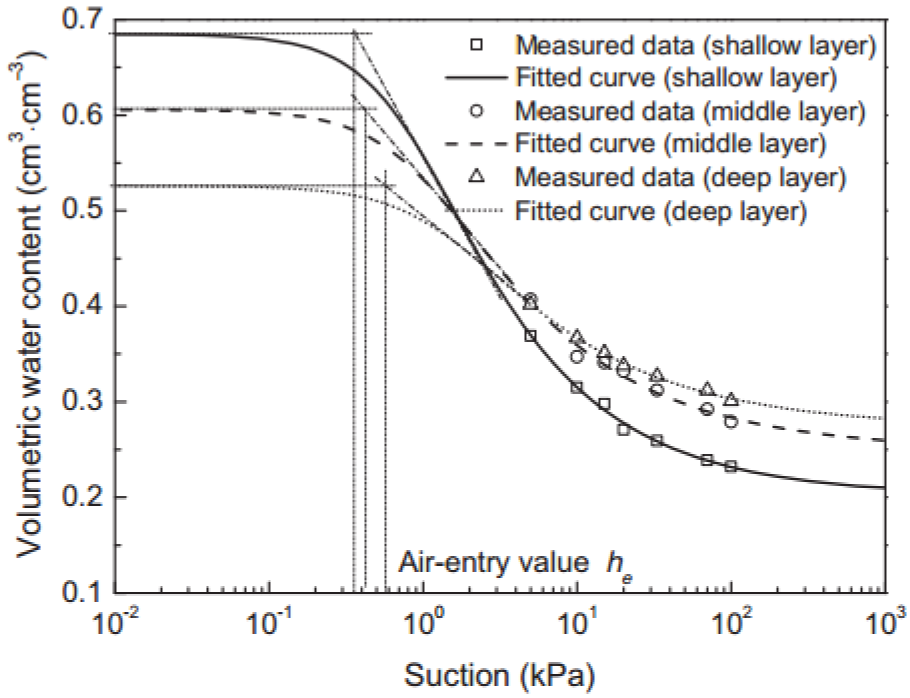


Figure 2.16 WRC for Three MSW Landfilled Samples (Wu et.al, 2012)

With having the van-Genuchten fitting parameters and the saturated hydraulic conductivity, the van Genuchten-Mualem function, as shown in Equation 4.3, can be used to predict the unsaturated hydraulic conductivity.

$$k(\psi) = k_s S_e^{0.5}\left[1-\left(1-S_e^{\frac{1}{m}}\right)^m\right]^2 \tag{4.3}$$

S_e= effective degree of saturation

Pressure plate test is a common test for obtaining the WRC. Unfortunately, the van-Genuchten fitting parameters for MSW are lacking. In the following paragraphs, the water retention curve characteristics for wastes in different literatures have been introduced.

Wu at al. (2011) carried out pressure plate tests on landfilled samples from a MSW landfill in Beijin, China. They collected samples from three different depths to evaluate the effects of age and depth on the unsaturated hydraulic properties of MSW. Table 2.14 and Table 2.15 show physical properties of the samples and the van-Genchten fitting parameters respectively.

Table 2.14 Physical Properties of MSW Samples Used for Pressure Plate Tests (Wu. et al., 2011)

	Shallow layer	Middle layer	Deep layer
Depth (m)	1-4	11-14	22-25
Age (years)	3	6	10
In situ unit weight (kN/m^3)	6.98	12.65	14.32
Gravimetric water content, Mc	0.27	0.45	0.35
Volumetric water content, θ_w	0.19	0.59	0.52

Table 2.15 The van-Genuchten Fitting Parameters (Wu. et al., 2011)

Sample	θ_r	θ_s	α (kPa^{-1})	n
Shallow Layer	0.2	0.69	1.18	1.59
Middle Layer	0.25	0.61	0.98	1.51
Deep Layer	0.27	0.53	0.71	1.49

In another research, Breitmeyer and Benson carried out hanging column test to estimate the parameters for the WRC (2011). They collected MSW samples with four months age from an operating landfill in Southern Wisconsin. They shredded the samples and then compacted them to three dry densities. Table 2.16 shows the van-Genuchten fitting parameters for the WRCs that they measured.

Table 2.16 The van-Genuchten Fitting Parameters (Breitmeyer & Benson, 2011)

Dry Density (Kg/m^3)	θ_r	θ_s	α (kPa^{-1})	n
561	0.21	0.60	3.38	1.89
632	0.22	0.53	2.92	1.6
795	0.03	0.41	1.18	1.33

Also, Kazimoglu, McDougal, and Pyrah studied the unsaturated hydraulic conductivity of the waste (2006). They carried out the pressure plate test on MSW samples from the UK. Table 2.17 shows the fitting parameters based on their study. They concluded that at low moisture content, the mechanism of flow in MSW is almost same with the mechanism in soil.

Table 2.17 The van-Genuchten Fitting Parameters (Kazimoglu, McDougal, & Pyrah, 2011)

θ_r	0.14
θ_s	0.58
α (kPa^{-1})	1.4
n	1.6

Table 2.19 summarizes the unsaturated hydraulic properties of MSW that are published yet. Also, Table 2.19 represents the unsaturated hydraulic properties of soil for 12 textural classes of USDA textural triangle. Table 2.12 and Table 2.19 both represents parameters for the analytical function of van Genuchten (1980).

Table 2.18 Published unsaturated hydraulic conductivity of MSW

Researchers	Landfill Location	Sample Condition	Dry Density (Mg/m^3)	θ_r	θ_s	n	α (1/m)
Stoltz et al. (2012)	France	Shredded-Fresh or very low degraded	0.46	0.2	0.69	3.23	30
			0.54	0.2	0.62	2.63	29
			0.62	0.2	0.58	2.38	2.3
			0.77	0.2	0.45	1.82	5.7
			0.46	0.35	0.63	2.7	20
			0.53	0.15	0.547	2.21	30
			0.58	0.15	0.52	2.08	20
			0.38	0.15	0.68	3.13	35
Khire and Saravanathiiban (2012)	USA	Fresh or Low degraded	0.69	0.26	0.66	1.4	50
			0.72	0.3	0.64	1.48	50
Breitmeyer and Benson (2011)	USA	Shredded-Fresh or Low degraded	0.56	0.21	0.6	1.89	33
			0.63	0.22	0.53	1.6	29
			0.79	0.03	0.41	1.33	12
Kazimoglu et al. (2011)	UK	Shredded-Fresh or Low degraded	-	0.14	0.58	1.6	14
Benson and Wang (1998)	USA		0.7	0.11	0.53	2.2	2.6
Wu et al. (2011)	China	3 year wastes	0.7	0.2	0.69	1.59	11.8
		6 year wastes	1.26	0.25	0.61	1.51	9.8
		10 year wastes	1.43	0.27	0.53	1.49	0.71

Table 2.19 Unsaturated hydraulic properties of soil (Schaap et al., 2001)

Textural class	θ_r $[L^3L^{-3}]$	θ_s $[L^3L^{-3}]$	α $[cm^{-1}]$	n [-]	K_s $[cm\ d^{-1}]$
Sand	0.053	0.375	0.035	3.18	643.
Loamy Sand	0.049	0.390	0.035	1.75	105.
Sandy Loam	0.039	0.387	0.027	1.45	38.2
Loam	0.061	0.399	0.011	1.47	12.0
Silt	0.050	0.489	0.007	1.68	43.7
Silty Loam	0.065	0.439	0.005	1.66	18.3
Sandy Clay Loam	0.063	0.384	0.021	1.33	13.2
Clay Loam	0.079	0.442	0.016	1.41	8.18
Silty Clay Loam	0.090	0.482	0.008	1.52	11.1
Sandy Clay	0.117	0.385	0.033	1.21	11.4
Silty Clay	0.111	0.481	0.016	1.32	9.61
Clay	0.098	0.459	0.015	1.25	14.8

Chapter 3

Experimental Program

3.1 Background

Characterization of landfilled MSW is an important consideration to make a realistic model of field condition. Leachate recirculation through MSW is strongly dependant on the MSW characteristics. To characterize the MSW, physical composition, moisture content, unit weight, permeability and volatile solids tests were carried out. The results were used to predict the leachate recirculation performance in the Cefe Valenzuela Landfill.

3.2 Selected Study Area

The Cefe Valenzuela Landfill , as shown in Figure 3.1, is located in Nueces County, 14 miles southwest of Corpus Christi’s City Hall, at the intersection of Farm to Market 2444 and County Road 20. The coordinates and elevation are: Latitude 27º 38' 12" N, Longitude 97º 34' 05" W, and Elevation 46.07 feet above Mean Sea Level (MSL).

The Cefe Valenzuela landfill property covers 2,273.59 acres. The layout includes the construction of two Type I municipal solid waste landfill units, each measuring approximately 810 acres. Currently, it received the permit from Texas Commission on Environmental Quality (TCEQ) to operate as ELR landfill.

Figure 3.1 The Cefe Valenzuela landfill

Landfilled MSW samples were collected from cell 3D and 4B, using 3 large diameter boreholes (BH-1, BH-2, and BH-3), as shown in Figure 3.2. 12 samples are collected from BH-1 and BH-2 from depth of 10 ft, 20 ft, 30 ft, 40 ft, 50 ft, and 60 ft. Other 4 samples are collected from BH-3 from depth of 10 ft, 20 ft, 30 ft, and 40 ft.

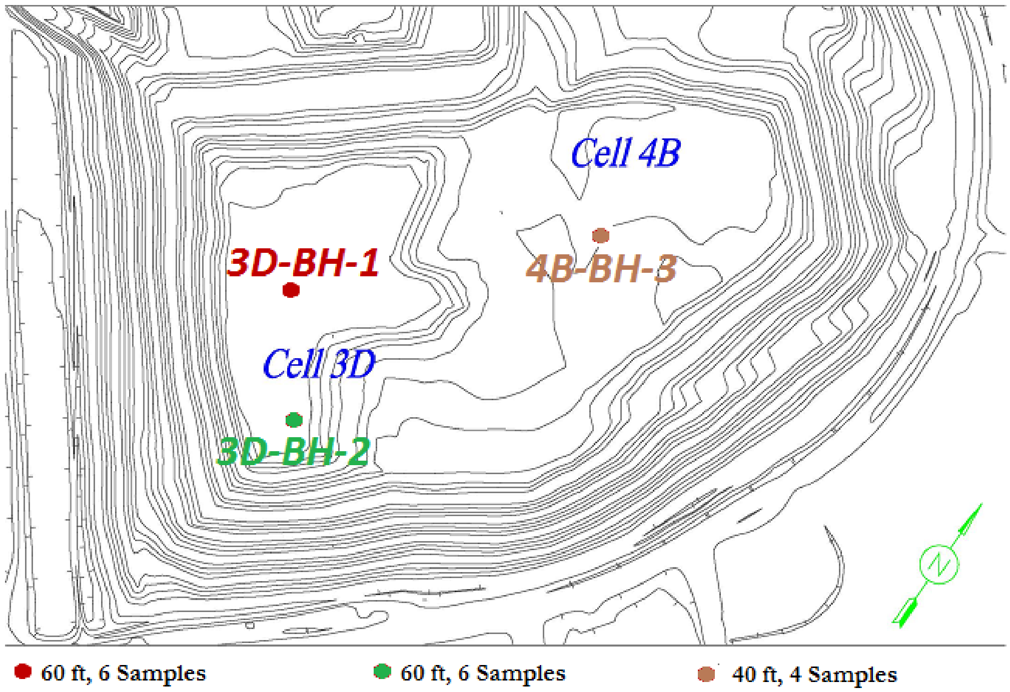


Figure 3.2 Locations of Boreholes

3.3 Sample Collection and Storage

Solid waste samples were collected from the Cefe Valenzuela landfill in May, 2013. A 3 ft diameter bucket augur attached to an AF130 Hydraulic Drill Rig was used for drilling, as shown in Figure 3.3. Solid waste samples were collected from 3 boreholes (3D-BH-1, 3D-BH-2, and 4B-BH-3).

Table 3.1 Sample collection details

Boring	**Location**	**Depth (ft)**	**Sampling**
BH-1	Cell 3D	60	6 samples, every 10 ft (10,20,30,40,50,60 ft)
BH-2	Cell 3D	60	6 samples, every 10 ft (10,20,30,40,50,60 ft)
BH-3	Cell 4B	40	4 samples, every 10 ft (10,20,30,40,50,40 ft)

(a) (b)

(c)

Figure 3.3 (a) AF 130 Hydraulic Drill Rig, (b) 3-ft Diameter Bucket Augur, and (c) Sample Collection

Total of 16 landfilled samples were collected for the experimental program. Samples were placed in sealed buckets to keep their moisture content, as shown in Figure 3.4. The weights of samples mostly varied from 15 lb to 20 lbs. The samples were brought to the laboratory and preserved at about 4^{o}C in environmental growth chamber, as shown in Figure 3.4.

(a) **(b)**

Figure 3.4 (a) Buckets of samples, and (b) Environmental growth chamber where the samples are kept

3.4 Experimental Program

An extensive experimental program was developed to charachterizes the MSW. The experimental program is presented in Table 3.2.

Table 3.2 Experimental program

Test	Sample	No. of Tests
Physical Composition	Landfilled MSW	6+6+4= 16
Moisture Content	Landfilled MSW	6+6+4= 16
Unit Weight (Standard Proctor Test)	Landfilled MSW	6+6+4= 16
Unit Weight (Compression Machine)	Landfilled MSW	6+6+4= 16
Hydraulic Conductivity	Landfilled MSW	6+6+4= 16
Volatile Solids	Landfilled MSW	6+6+4= 16

The methodologies that have been used to conduct these tests are described in the following subsections.

3.4.1 *Physical Composition*

To conduct the physical composition test, first the content of a bucket was poured in a big tray. Then the constituents of the waste mass is manually separated to paper, plastic, food waste, leather & textile, wood & yard waste, metals, glass, styrofoam & sponge, construction\demolition (C & D) debris and others. Figure 3.5 shows the separated constituents after sorting.

Figure 3.5 Seperation of an MSW mass to its constituents

The paper category includes all kinds of papers like cardboard packaging, newspaper, magazines, office papers, etc. All plastic polythene bags, containers, food wrappers, plastic bottles and rubber objects were classified as plastic. All clothes, fabrics, leathers, etc., and categorized as leather & textile. Branches, leaves & grass from garden trimming, and also broken pieces of wood from construction & demolition categorized as wood and yard waste. All metal cans, cutlery and food container were placed under metal category. Construction debris constituted of limes, bricks and stone chips, broken tiles, the construction insulation material, etc. Whatever could not be classified, like fine particles and soil, were classified as others.

After the completion of separation, every category was weighted and the weight presented as the percentage of total weight. Paper, food waste, leather and textile, and wood and yard waste categories were considered degradable and the other categories except others considered as non-degradable categories. Others category was divided to degradable and non-degradable portion based on the VS value.

3.4.2 *Moisture Content*

Moisture content of samples were determined based on standard methods of ASTM D 2974 – 00 and APHA 2540 – B. First, a representative sample, weighting 1.5-2 lb, from each bucket was used to determine the moisture content. Then the weight of the sample was recorded before drying. Next, the sample was dried at 105°C in the oven for 24 hours. The dried sample was weighted and finally the moisture content was calculated. Moisture content can be expressed in two ways, wet weight (w/w) basis and dry weight basis (d/w). Equations 3.1 and Equations 3.2 were used to determine moisture content on wet weight basis and dry weight basis, respectively. Figure 3.6 shows sample being dried in the oven for the determination of moisture content.

$$Moisture\ content, \%\ (wet\ wt\ basis) = \frac{a-b}{a} \times 100 \tag{3.1}$$

Where, a = initial weight of the sample as delivered; and

b = weight of the sample after drying

$$Moisture\ content, \%\ (wet\ wt\ basis) = \frac{a-b}{b} \times 100 \tag{3.2}$$

Where, a = initial weight of the sample as delivered; and

b = weight of the sample after drying

Volumetric moisture contents have been estimated, using the Equation 3.3.

$$\Theta = \frac{w_d * \gamma}{(1 + w_d) * \gamma_w} \tag{3.3}$$

w_d=dry gravimetric moisture content

γ= unit weight of the waste sample

γ_w=unit weight of water

Figure 3.6 Drying the Samples for Moisture Content Test

3.4.3 *Unit Weight*

Unit weight of MSW samples are calculated based on two methods: The first one is the unit weight using standard proctor compaction, ASTM D698, and the second one is the unit weight after the application of a specified overburden pressure. Both methods are conducted on samples with their natural moisture content.

For standard proctor compaction, because of larger particle size, a bigger mold than the conventional mold size for soils was used. The size of mold is 6 inch in diameter and 6.1 inch in height which makes a volume of 0.1 ft^3. Waste materials are placed in three layers and each layer is compacted before the placement of the next layer. Figure

3.7 shows a sample while it is being compacted. For compaction, a 5.5 lb was dropped 75 times for a fall height of 12 inch in each layer to apply the standard compaction effort to all the volume. The application of 75 blows instead of conventional 25 blows for each layer is due to the greater volume of the mold. For the tests the standard hammer is used, so P an h in Equation 3.4 are same with the standard conventional method. As the volume of the mold for these tests are three times bigger than the conventional mold, n for the bigger mold should be 3 time of the n for the small mold, which is 25 blows, for keeping the compaction effort same as the standard proctor compaction effort.

$$Compaction\ Effort, E = n.h.\frac{P}{V} \tag{3.4}$$

$n = number\ of\ blows, h = fall\ height$

$P = weight\ of\ hammer, V = volume\ of\ the\ mold$

Figure 3.7 Applying the standard proctor compaction effort

After compaction, the sample is weighted and with the known volume unit weight can be calculated based on Equation 3.5.

$$Unit\ Weight = \frac{weight\ of\ the\ compacted\ sample\ in\ the\ mold\ (lb)}{volume\ of\ the\ mold\ (ft^3)} \tag{3.5}$$

In the second method, the unit weight was determined using overburden pressure. This method is used to simulate the field overburden pressure applying on the sample and make the unit weight of sample close to whatever it is in the field. Overburden pressures at different depth are calculated based on an assumed unit weight. In the Cefe Valenzuela landfill the commonly compactor provides 900-1000 lb/yd^3 compaction. Consequently, the unit weight of 35 lb/ft^3 is assumed for the wastes and unit weight of 120 lb/ft^3 is assumed for 1 ft cover soil layer. The 60 kPa tensile-compression machine is used, as shown in Figure 3.8 (a), to compact the samples under the calculated overburden pressure corresponding to the depth of the sample. After compaction Equation 3.5 can be used to calculate the unit weight.

(a) **(b)**

Figure 3.8 (a) Applying the overburden pressure and (b) the compacted sample

3.4.4 *Hydraulic Conductivity*

Hydraulic conductivity test is conducted according to ASTM D 2434-68, Permeability of Granular Soils (Constant Head). The sample first is compacted by the 60 kPa compression machine under the specified overburden pressure, to gain more realistic unit weight. Unit weight of the sample is one important parameter affecting the hydraulic conductivity value. After compaction, the mold top cap is placed and the mold is closed in a way to avoid any leakage.

Figure 3.9 (a) shows the test setup. Once the sample inside the mold is saturated, by recording the discharged water from water outlet with its corresponding time hydraulic conductivity can be determined based on Equation 3.6.

$$k = \frac{Q.L}{A.t.h} \tag{3.6}$$

K= coefficient of permeability in cm/s

Q= quantity of waster discharged in cm3

L=distance between manometers opening in cm

T= total time to collect the quantity Q seconds

H= difference in head on manometers in cm

A= cross-sectional area of specimen in cm2

D= Inside diameter of permeameter in cm

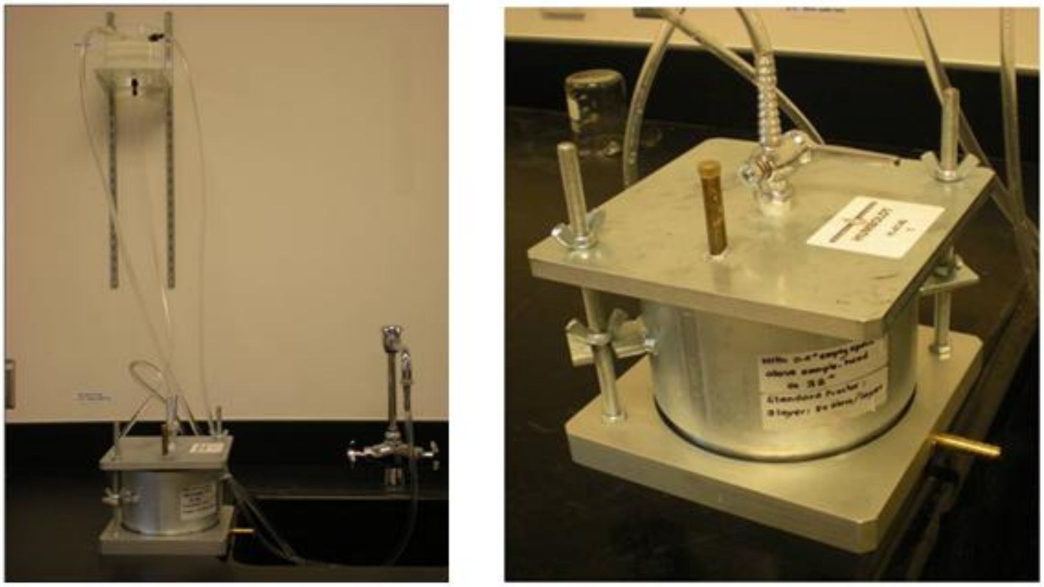
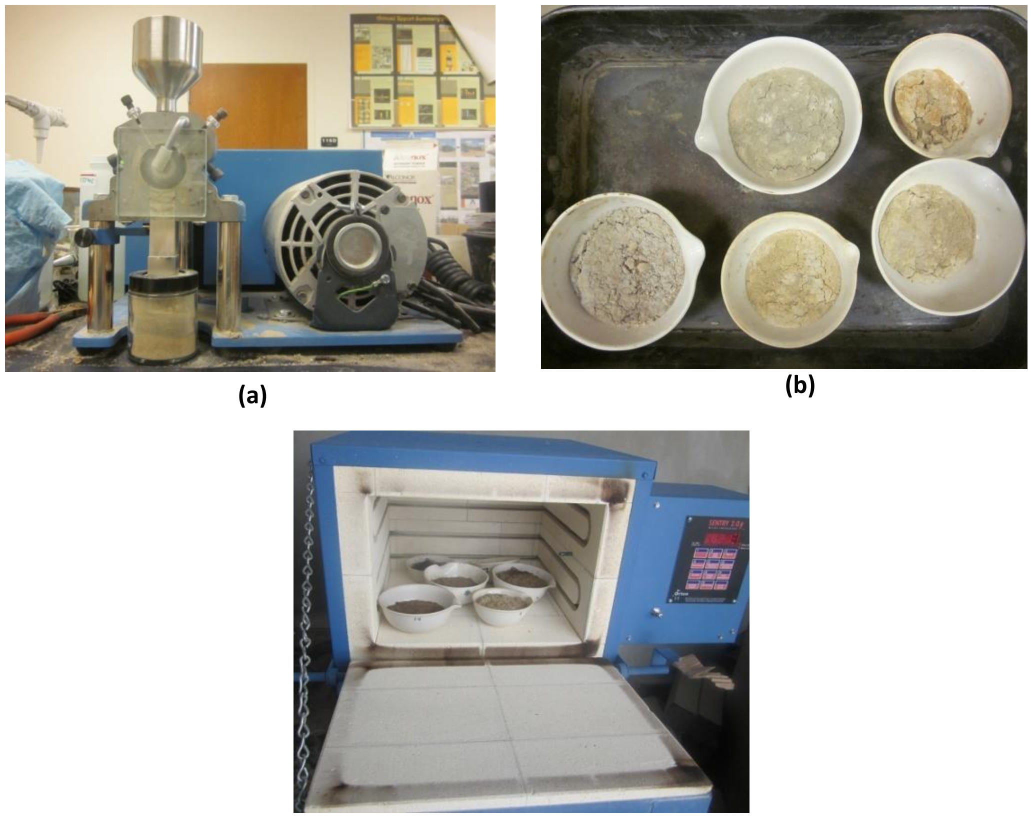


Figure 3.9 (a) Entire hydraulic conductivity setup and (b) Hydraulic conductivity mold

3.4.5 *Volatile Solids*

Volatile solids test is used to evaluate the content of remained biodegradable materials in a waste mass. The volatile solids procedure followed a modified version of Standard Methods APHA Method 2440-E. Samples were dried once again at 105°C to a constant weight and held in a desiccator. Approximately 50 grams of dried and grinded MSW were placed in pre-weighed porcelain crucibles and inserted into a muffle furnace at 550°C for 2 hrs.



(a) (b)

(c)

Figure 3.10 (a) Shredding the Oven-Dried Sample (b) Samples after Ignition, and (c) Placement of Samples in the Furnace

Chapter 4

Numerical Modeling Methodology

4.1 Background

Developing a conceptual model for numerical simulations that reflects the actual condition of the field is an important step to obtain a valid result. Laboratory tests were helpful to predict the MSW properties. Also, the simulations for the City of Denton Landfill confirmed the selected methodology of this study.

In this study, the modeling methodology is developed based on the HYDRUS computer model.

4.2 HYDRUS 2D computer model

Hydrus-2D is a finite element and windows based software that can simulate waster, heat, and solute movement in unsaturated, partially saturated, or fully saturated porous media. The program numerically solves Richards' equation for saturated/unsaturated water flow. Korfiatis et al. (1984) showed the validity of the Richard's equation for predicting the saturated and unsaturated flow in MSW samples. Equation 4.1 represents a 2D from of Richards' equation.

$$\frac{\partial\theta}{\partial t} = -\frac{\partial}{\partial x}\left[k(\psi)\ \frac{\partial\psi}{\psi x}\right] - \frac{\partial}{\partial z}\left[k(\psi)\ \frac{\partial\psi}{\psi z}\right] + \frac{\partial k(\psi)}{\partial z} - S \tag{4.1}$$

Where:

Θ = Volumetric water content [dimensionless]

Ψ =Metric Suction head [L]

k = Hydraulic conductivity of the porous material which is dependent on the pressure head or water content [L/T]

S = Volume of water removed per unit time per unit volume of soil by plant water uptake or evaporation [1/T]

t = Time [T]

In this study, HYDRUS-2D was used to simulate the leachate recirculation using vertical wells in the Cefe Valenzuela Landfill. Beside many capabilities of this computer model, it was found user friendly. In this software, various types of boundaries, such as constant or time-variable prescribed head, flux, or controlled by atmospheric conditions, can be selected. Also, the radial vertical flow option that the software provides was found useful for the purpose of this study which is molding of leachate injection using vertical wells.

Khire and Mukherjee (2006) used HYDRUS-2D to model leachate injection using vertical wells in bioreactor landflls. In older studies, McCreanor and Reinhart (1996), and Jain (2005) used SATURA to simulate the saturation profiles around a vertical well after the leachate injection.

4.3 Boundary Conditions

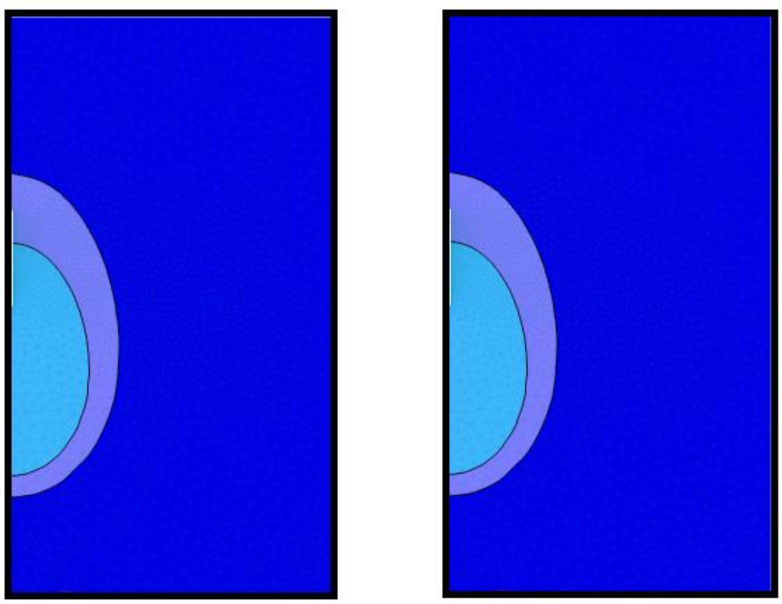
HYDRUS-2D provides various boundary conditions. In the current study, the bottom level of the domain is assumed as a free drainage boundary. This assumption is controlled by modeling a harsh condition.

Other external lines are assumed as no-flux boundaries. Although the assumed domain is not completely isolated in reality, interaction of the domain with its surroundings through processes such as evapotranspiration is negligible.

The well surface has been assigned as a flux boundary. The desired flux value can be allocated to this boundary. According to the literature, once the hydraulic conductivity of the drainage pack is greater than that of wastes, the drainage pack has no effect on the leachate extent. Thus, the flux boundary has been assigned to the outside surface of the drainage pack.

4.4 Modeling Parameters

Design parameters are the parameters that affect the results. The important design parameters are waste flow properties, leachate quantity, time of injection, and the well dimensions. Leachate quantity and time of injection have been selected based on the common field practices. In this study a wide range of leachate quantity has been assumed based on common field practices. The injection time can be less than one hour to a couple of hours. In this study, the leachate injection time is assumed 4 hours. According to Figure 4.1, time of injection in a range of few hours does not affect the results significantly.



(a) (b)

Figure 4.1 Saturation profile at day 50 after injection of 4000 gallon in a) 2 hours, and b) 4 hours

Well dimensions including the diameter and the height are also assumed based on the preliminary design phase details. All the wells have a 3 m perforated section at their bottom. The wells diameter is 0.15 m including the drainage pack.

The most important parameter is the waste properties. Although there are some literatures about the waste flow properties, they cannot directly be used as the waste properties because of specific conditions of the waste samples. The complexity in the nature of the waste materials and the difficulty of the tests that measure the unsaturated flow properties of the wastes are the major reasons that why so far no attempt to generalize the waste flow properties has been done. In this study, waste flow properties have been chosen based on the literatures and the author's judgment. For the key simulations, two set of material properties have been used to represent the fresh and degraded MSW.

4.5 Material Models

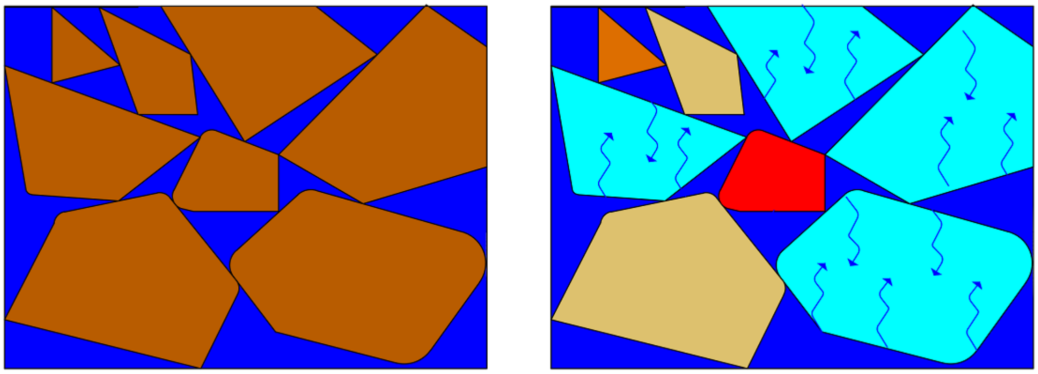
For this study two sets of materials as MSW have been selected. The properties are shown in Table 4.1. MSW unsaturated properties have been selected based on the results in Table 2.18 and simulations for the City of Denton Landfill. The available electrical resistivity imaging data after leachate injection for this landfill allows the approximate validation of waste properties. Generally, the wetter zone has low electrical resistivity. Therefore the difference in electrical resistivity of wastes before and after leachate injection shows the wetted area. Although the recirculation injection in this landfill is through horizontal trenches and permeable blankets, it is still helpful for validating the waste flow properties.

Table 4.1 MSW flow properties used as MSW

MSW Condition	θ_r	θ_s	α (kPa^{-1})	n	K_S (cm/s)
Fresh	0.20	0.58	3.5	1.75	1.2×10^{-2}
Degraded	0.22	0.43	0.5	1.66	2×10^{-4}

The residual volumetric water content, θ_r, is selected based on the published values for wastes in literatures. Indeed, using the soil-like value because the different nature of the waste materials is not correct. Comparison of θ_r values for soils and wastes that are shown in Table 2.18 and Table 2.19 confirms this point. The average θ_r for fresh or low degraded wastes based on values mentioned in Table 2.18 is 0.2; however, the average value for soil based on values mentioned in Table 2.19 equals to 0.07. The main reason for this difference is the different texture of soil particles and waste materials. Indeed, as it is shown in Figure 4.2, a significant part of MSW such as paper constituents can absorb water and retain moisture internally. Obviously the trapped moisture inside the very tiny pores of constituents such as papers increases the value of θ_r because very high matric suction is required to remove this moisture.

As the MSW degrades it includes more fine particles so θ_r increases. θ_r for degraded materials has been selected as 0.22. While this value reflects the more degraded state of the waste, still moisture should be added to reach the optimum moisture content. The initial water content of the domain is also assumed same as θ_r.

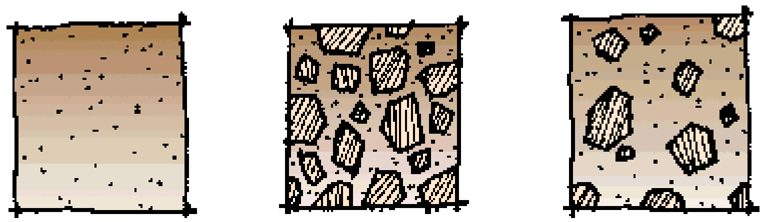


(a) (b)

Figure 4.2 Demonstration of a) Soil matrix, b) Waste matrix

The saturated volumetric water content, θ_s, is also selected based on the published values for wastes in literatures. The average θ_s for fresh or low degraded wastes based on values mentioned in Table 2.18 is 0.58; however, the average value for soil based on values mentioned in Table 2.19 equals to 0.43. Although the compaction effort is an important parameter, the main reason of higher porosity in wastes is the particle size distribution. MSW include different constituents including bulky items which increases the voids. As the MSW degrades, it includes finer particles and porosity or θ_s decreases. θ_s might be lower than porosity because landfill gas can occupy a portion of voids.

For degraded MSW, the θ_s has been selected as the average θ_s of those samples from the USA or Europe with higher dry density in Table 2.18. Therefore, the θ_s values of 0.45 and 0.41 which are corresponding to dry densities of 0.77 and 0.79 Mg/m^3 have been considered as the representing values with the average of 0.43. This is same as the silt loam value that has been used in other studies.



(a) (b) (c)

Figure 4.3 Demonstration of a) Clay and silt, b) Loam, and c) Sandy soil texture (City of Bremerton Website, 2013)

Fitting parameters of the van Genuchten function of n and α were assumed similar to the fitting parameters of two USDA soil textures. Figure 4.3 depicts three general textures of soil. Among these textures loamy texture is better characterizing the MSW texture. Loam is a mix of sand, silt or clay, and organic matter. Loamy soils generally have a good moisture absorption and retention capacity.

For fresh or low degraded MSW, n and α are assumed similar to loamy sand values. This texture contains lower portion of fine particles among the larger particles. n and α of this texture are relatively high which means it is more drainable texture. On the other hand, for degraded materials the silty loam parameters are selected. Silty loam contains greater portion of fine particles in the soil texture. This texture is also recommended by literature for degraded MSW. α of this texture is relatively low which means it is a less drainable texture.

Table 4.2 Selection of n and α from unsaturated hydraulic properties of USDA soil textures (Schaap et al., 2001)

Textural class	θ_r [L^3L^{-3}]	θ_s [L^3L^{-3}]	α [cm^{-1}]	n [-]	K_s [$cm\ d^{-1}$]
Sand	0.053	0.375	0.035	3.18	643.
Loamy Sand	0.049	0.390	0.035	1.75	105.
Sandy Loam	0.039	0.387	0.027	1.45	38.2
Loam	0.061	0.399	0.011	1.47	12.0
Silt	0.050	0.489	0.007	1.68	43.7
Silty Loam	0.065	0.439	0.005	1.66	18.3
Sandy Clay Loam	0.063	0.384	0.021	1.33	13.2
Clay Loam	0.079	0.442	0.016	1.41	8.18
Silty Clay Loam	0.090	0.482	0.008	1.52	11.1
Sandy Clay	0.117	0.385	0.033	1.21	11.4
Silty Clay	0.111	0.481	0.016	1.32	9.61
Clay	0.098	0.459	0.015	1.25	14.8

The saturated hydraulic conductivity (k_s) has been assumed based on the previous studies and the test results. For the degraded material, the minimum value of permeability from the laboratory test results has been selected. The minimum Ks value has been recorded for BH-2 at 10 ft, which is 2×10^{-4}. For the fresh material, the K_s has been assumed based on values in Table 4.3 Selection of Ks for fresh MSW base on the published values, because the similarities in physical composition and the dry unit weight. Based on the test results the average unit weight of the samples is 0.5 Mg/m_3.

Table 4.3 Selection of Ks for fresh MSW base on the published values

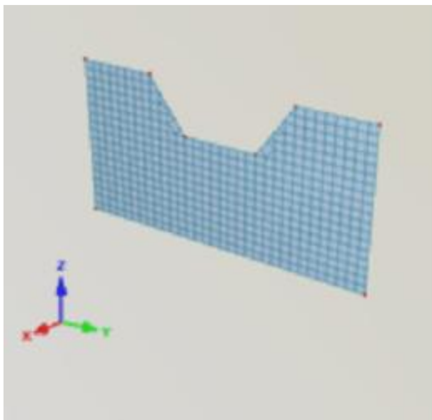
Source	Testes MSW	Dry Density ρ_d Mg/m^3	Compression σ' (kPa)	Hydraulic Conductivity (cm/s)
Durmusoglu et al. (2006)	MSW from Texas	-	123-369	$1.2 \times 10^{-2} - 4.7 \times 10^{-4}$
Reddy et al. (2009)	MSW from Illinois	0.41-1.33	0-276	$2 \times 10^{-1} - 4.9 \times 10^{-5}$
Stoltz et al. (2010)	MSW from France	0.32-0.96	0-276	$2 \times 10^{-1} - 7.8 \times 10^{-5}$
		0.36-0.6	0-200	$1.6 \times 10^{-1} - 4.9 \times 10^{-4}$
		0.49-0.6	100-200	$1 \times 10^{-2} - 1.1 \times 10^{-3}$

4.6 Conceptual Models and Assumptions

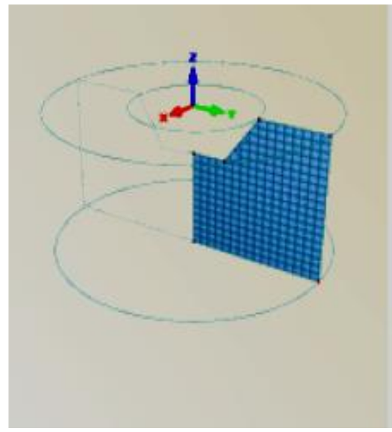
Developing the conceptual model for numerical simulations that represents the field condition is an important step. Here, it has been tried to develop the model that represents the condition as close as possible to the actual field condition. Meanwhile, the simplifications and assumptions are inevitable to develop an efficient and functional model.

In this study, HYDRUS-2D has been used for modeling the leachate recirculation using horizontal trenches and vertical wells. For modeling the horizontal trench the vertical plane domain, as shown in Figure 4.4 (a), has been used because the flux boundary geometry is same cross plane. However for modeling the vertical wells

HYDRUS-2D provides a very handy option as axisymmetrical vertical flow domain, as shown in Figure 4.4 (b). This option is useful to model the vertical radial flow, which is the case here.



(a)



(b)

Figure 4.4 Hydrus-2D domain options a) Vertical plane and b) Axisymmetrical vertical flow

Once the domain has been selected, the next important step is spatial and temporal discretization. In order to solve the non linear Richard's Equation, the software needs appropriate spatial and temporal discretization. This is a crucial step to develop the model that works efficiently. Smaller mesh sizes prolongs the calculation time, however, the model may not work using larger sizes. Therefore, the optimum mesh sizes should be obtained by some preliminary model calculations. Also, the mesh sizes have been decreased around the well, where there is a large head pressure and changes in the domain.

Temporal discretization or time step is also important in order to run the model successfully. Temporal discretization mainly depends on the soil texture, leachate pressure head and the mesh sizes. Δt_{ini} , and Δt_{min} are two important input parameters that HYDRUS requires to solve the equation. Δt_{ini} or the initial time step is the time step that the software uses to begin the calculation. If this time step was not small enough, the software automatically reduces the time step. The minimum possible time step is Δt_{min} . According to the HYDRUS user manual, a large Δt_{ini} value is better to be used for soil with less nonlinear hydraulic properties such as loam. MSW has also loam-like properties. Also, very small Δt_{min} has been used in simulations because of the significant pressure head changes at the flux boundary.

Figure 4.5 shows a meshed axisymmetrical radial and vertical flow domain. The width of the domain has been inserted beyond the affected zone. The height of domain is as the actual landfill height, which is 18 m or 60 ft. Figure 4.6 and Figure 4.7 depict the geometry of the conceptual models for single well and well cluster arrangements.

For the single well analysis the bottom level of the well is at 9 m or 30 ft. This level is far away enough from the bottom drainage layer to see the maximum affected zone.

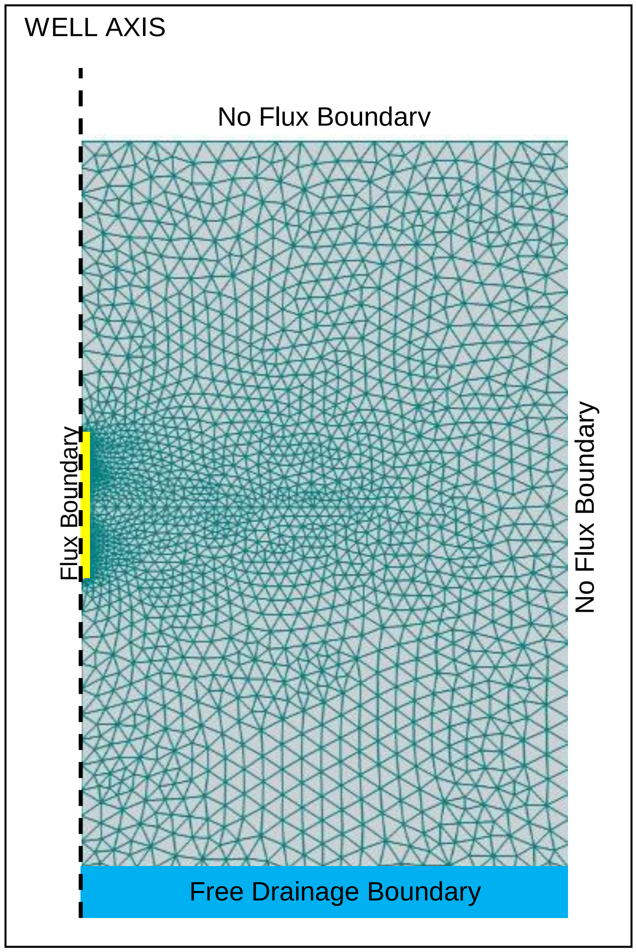


Figure 4.5 Axisymmetrical vertical flow meshed domain

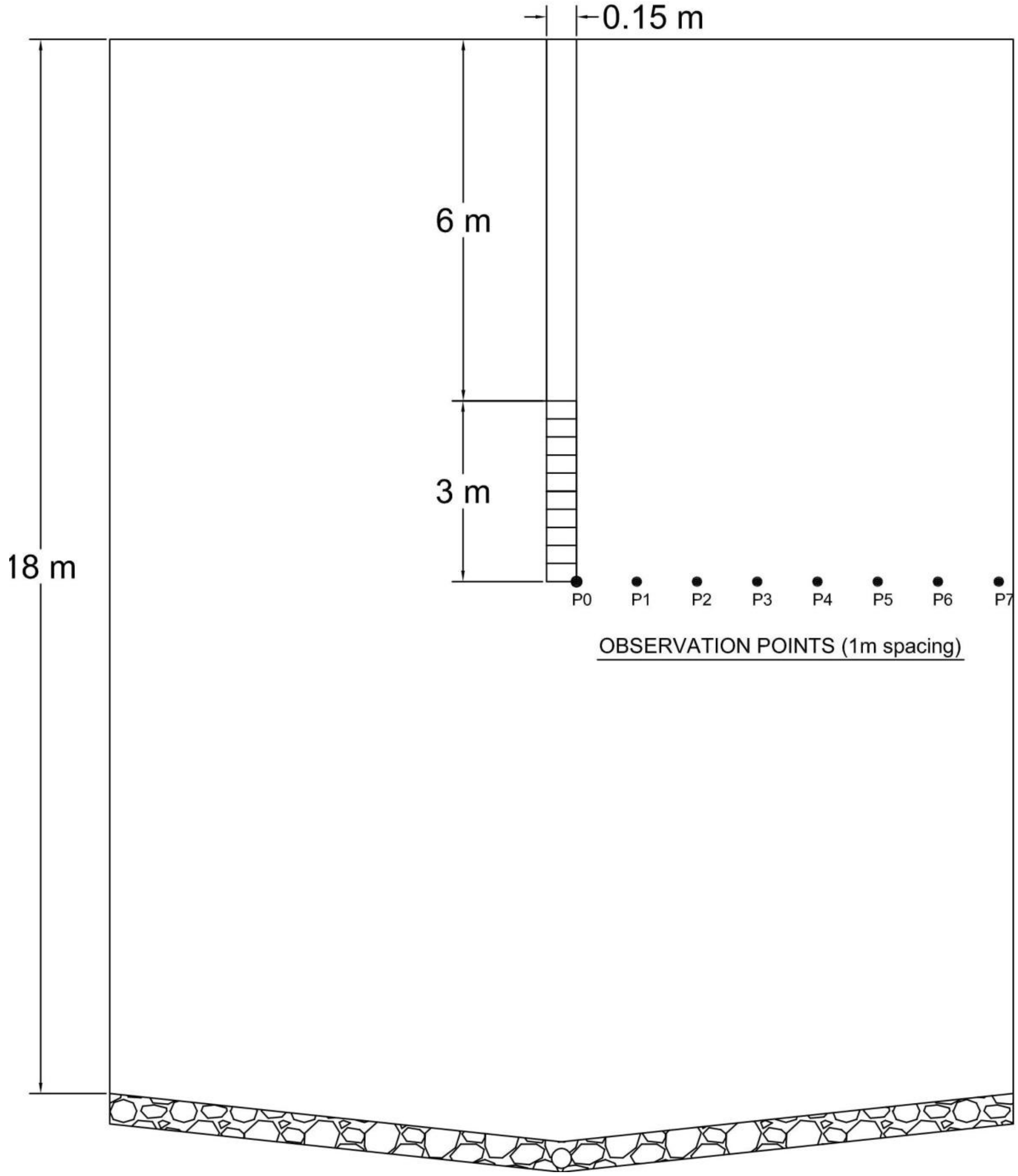


Figure 4.6 Model geometry for single wells

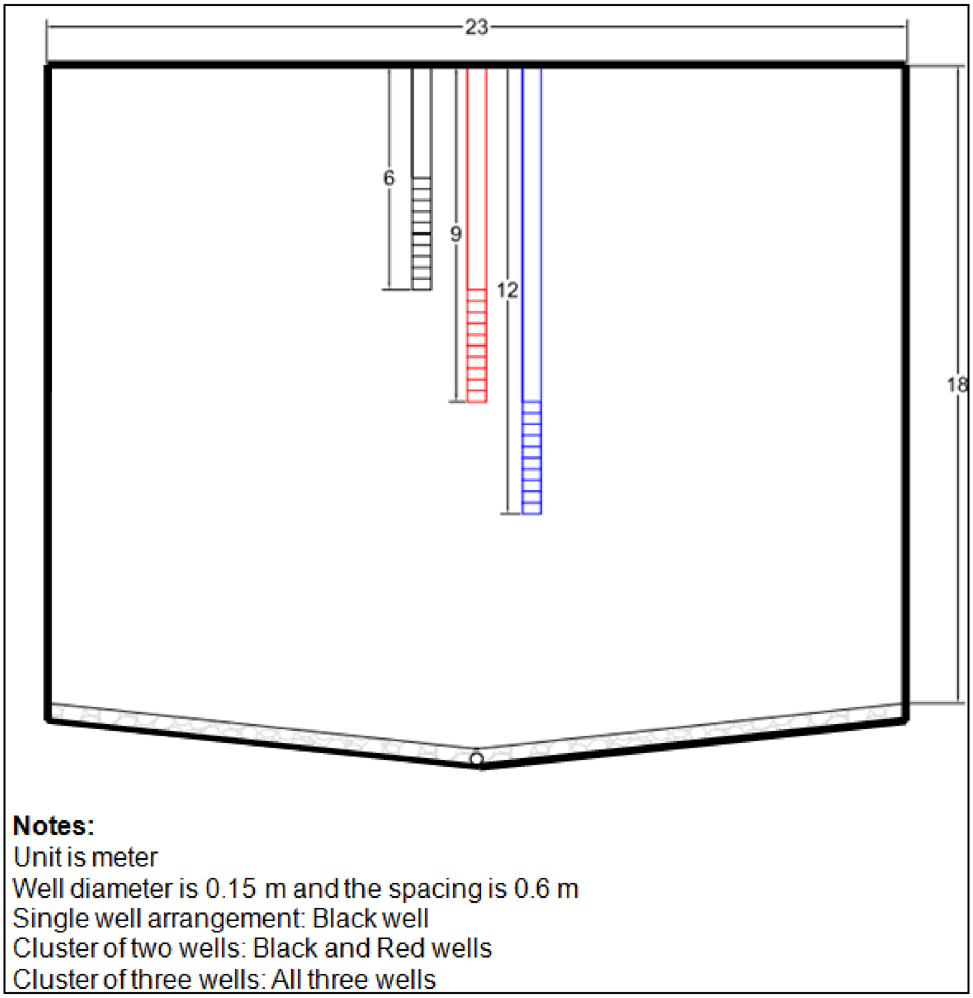


Figure 4.7 Model geometry for single and cluster arrangements

Chapter 5

Results and Discussions

5.1 Introduction

5.2 Laboratory Test Results

5.2.1 *Physical Composition*

Physical composition tests have been done to confirm the potential of the landfill for gas generation. Degradable organics are degraded by microorganisms, resulting gas generation. The greater degradable organics content of the MSW in the landfill results the greater gas generation capacity. Therefore, it's important to estimate the degradable portion of the MSW. Figure 5.1, Figure 5.2, and Figure 5.3 show the physical composition of the samples. Figure 5.4 shows the average weight percentages of MSW from all samples. Figure 5.5 represents the degradable and non degradable portion of the MSW for the landfill. Fine particles have been assumed as non degradable constituents that is a more conservative assumption.

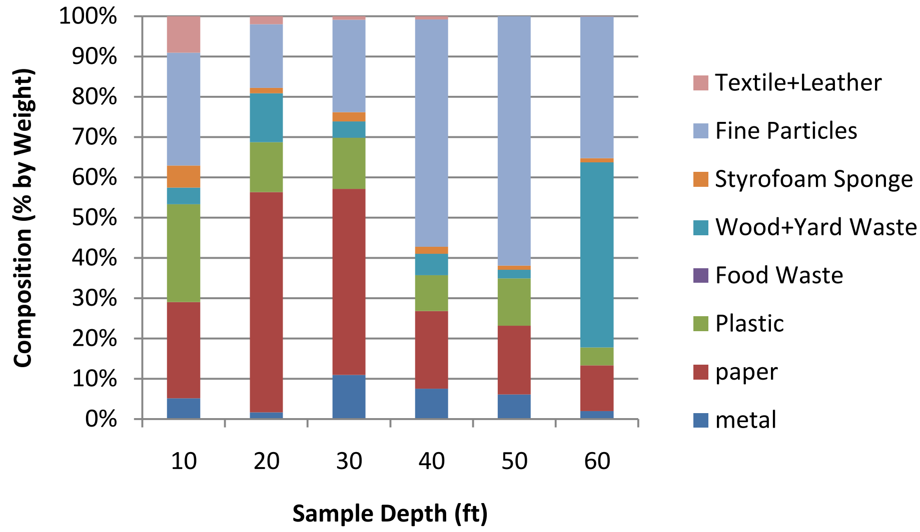


Figure 5.1 Physical Composition of samples from BH-1

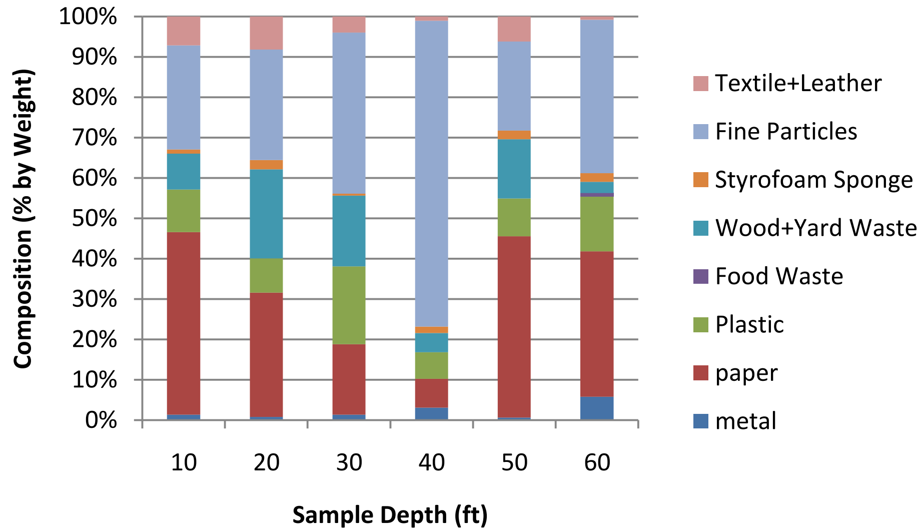


Figure 5.2 Physical Composition of samples from BH-2

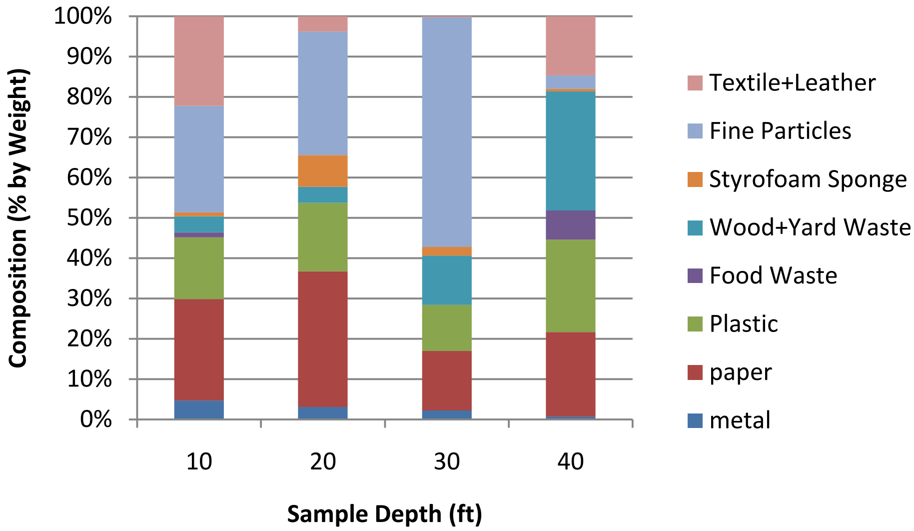


Figure 5.3 Physical Composition of samples from BH-3

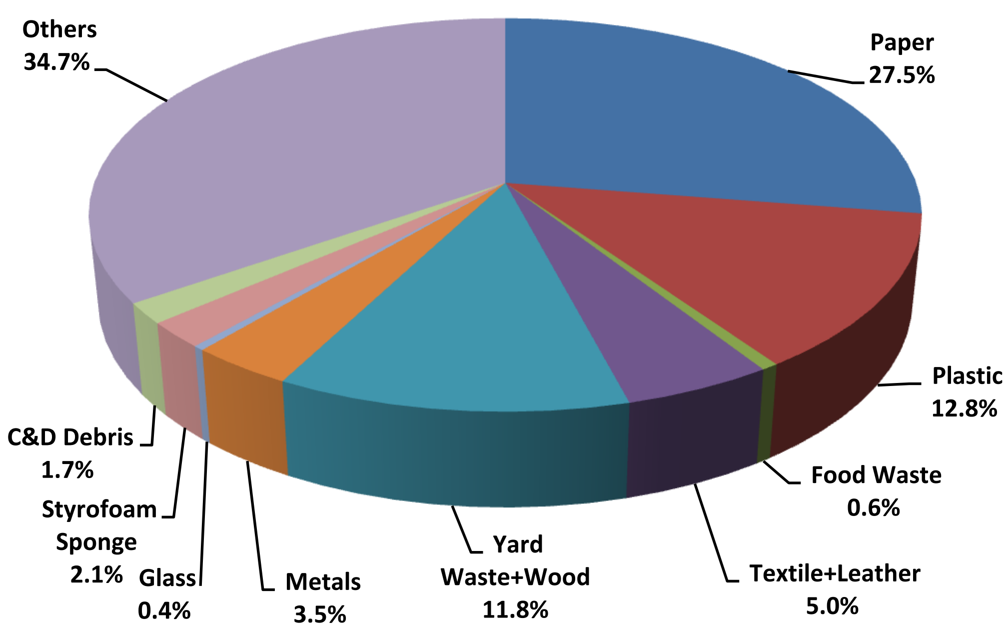


Figure 5.4 Average weight percentages of MSW from all samples

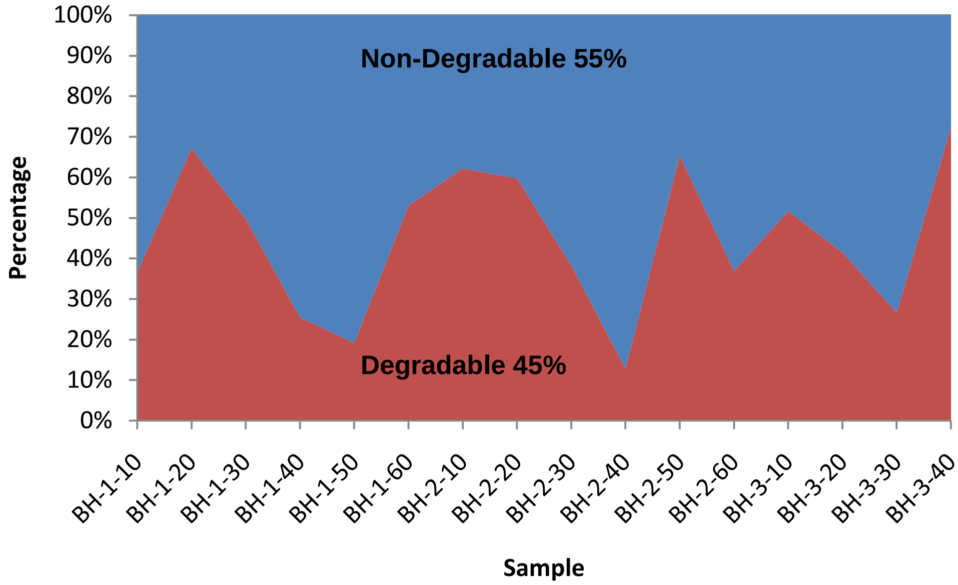


Figure 5.5 Degradable and non-degradable portion of the wastes

5.2.2 *Moisture Content*

Moisture condition tests have been performed for determining the initial moisture condition of wastes. Figure 5.6 and Table 5.1 demonstrates the gravimetric moisture content of the samples. Volumetric water contents are shown in Table 5.2.

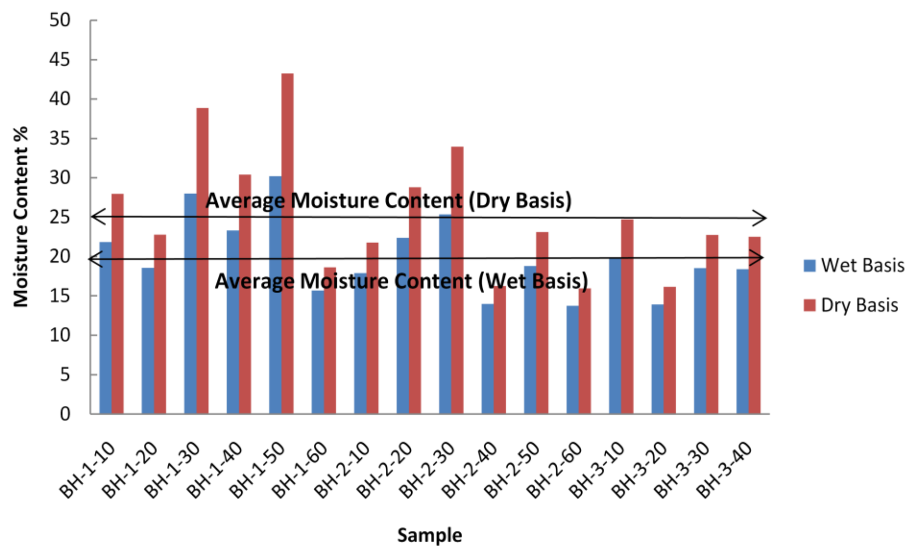


Figure 5.6 Moisture content of samples

Table 5.1 Moisture contents

Depth (ft)	BH-1		BH-2		BH-3	
	MC (wet weight basis) (%)	MC (Dryweight basis) (%)	MC (wet weight basis) (%)	MC (Dry weight basis) (%)	MC (wet weight basis) (%)	MC (Dry weight basis) (%)
10	21.84	27.95	17.89	21.79	19.83	24.73
20	18.55	22.78	22.36	28.80	13.91	16.16
30	27.99	38.87	25.35	33.96	18.52	22.73
40	23.31	30.40	13.99	16.26	18.38	22.52
50	30.19	43.24	18.78	23.12	-	-
60	15.70	18.62	13.76	15.95	-	-
Average	22.93	30.31	18.69	23.31	17.66	21.53
Avg. MC wet weight basis (%)			19.76			
Avg. MC dry weight basis(%)			25.05			

Table 5.2 Estimated volumetric water contents

Boring	Depth (ft)	Volumetric water content
3D-BH-1	10	13
	20	10
	30	16
	40	14
	50	29
	60	6
3D-BH-2	10	9
	20	10
	30	16
	40	11
	50	15
	60	10
4B-BH-3	10	10
	20	9
	30	15
	40	7
Average		12.5
Standard Deviation		5.39
Maximum		29
Minimum		6

5.2.3 *Unit Weight*

Unit weight is one the most important engineering properties of the waste because it’s in correlation with many properties such as compaction quality, degradation state, permeability, and unsaturated flow properties. Table 5.3 and Figure 5.7 show the unit weights after application of the standard proctor effort. Table 5.4 and Figure 5.8 show the unit weights after application of overburden pressure. In comparison with Table 2.9 and Table 2.10, it is obvious that unit weights are in a range of fresh waste materials.

Table 5.3 Unit weights of samples (Standard Proctor)

Boring	Depth (ft)	Compacted Density (pcf)	Compacted Unit Weight (kN/m^3)
BH-1	10	50.52	8.09
	20	33.79	5.41
	30	35.12	5.63
	40	38.52	6.17
	50	58.72	9.41
	60	25.79	4.13
BH-2	10	33.13	5.31
	20	27.39	4.39
	30	34.86	5.58
	40	62.45	10
	50	42.72	6.84
	60	39.79	6.37
BH-3	10	35.92	5.75
	20	37.72	6.04
	30	65.65	10.52
	40	21.93	3.51
Average		40.25	6.45
Standard Deviation		12.85	2.06
Maximum		65.65	10.52
Minimum		21.93	3.51

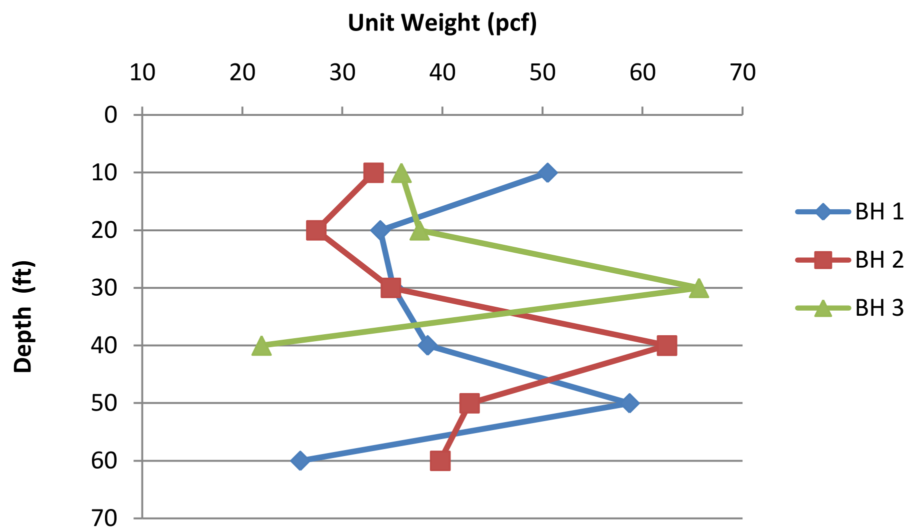


Figure 5.7 Variation of unit weight (Standard Proctor) with depth

Table 5.4 Unit weights of samples (Compression Machine)

Boring	Depth (ft)	Unit Weight (pcf)
BH-1	10	36.53
	20	32.29
	30	35.55
	40	37.18
	50	60.13
	60	25.15
BH-2	10	29.8
	20	28.88
	30	38.29
	40	49.11
	50	48.76
	60	44.07
BH-3	10	31.88
	20	41.97
	30	51.24
	40	24.66
Average		38.47
Standard Deviation		10.09
Maximum		60.13
Minimum		24.66

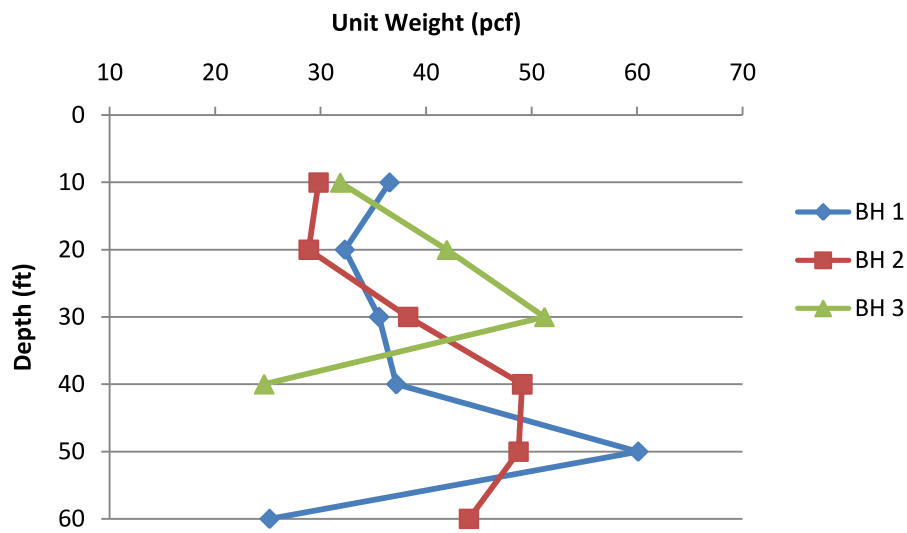


Figure 5.8 Variation of unit weight (Compression Machine) with depth

5.2.4 *Hydraulic Conductivity*

Saturated hydraulic conductivity of the samples is one the most important factors for leachate flow prediction. Table 5.5 and Figure 5.9 show the permeability of the waste at different depths.

Table 5.5 Permeability of samples

Boring	Sample No.	Permeability (cm/sec)
BH-1	10	2.69E-04
	20	3.38E-03
	30	2.70E-03
	40	2.24E-04
	50	4.60E-04
	60	3.73E-04
BH-2	10	1.97E-04
	20	7.60E-04
	30	3.74E-04
	40	4.60E-04
	50	8.49E-04
	60	3.27E-04
BH-3	10	6.68E-04
	20	4.33E-04
	30	4.12E-04
	40	2.55E-03
Average		9.03E-04
Standard Deviation		1.01E-03
Maximum		3.38E-03
Minimum		1.97E-04

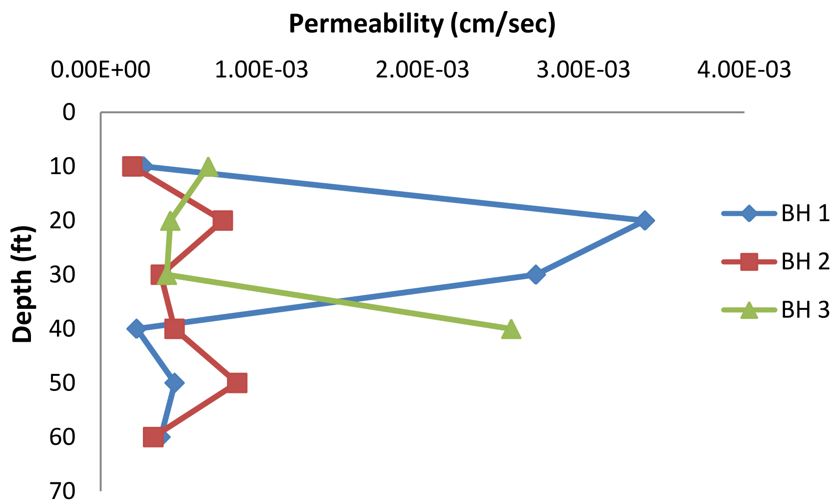


Figure 5.9 Permeability of landfilled MSW samples with depth

5.2.5 *Volatile Solids*

Volatile solid tests have been done to evaluate the degradation state of the wastes and the potential for gas production in future. Figure 5.10and Table 5.6 show the volatile solid percentage of samples at different depths.

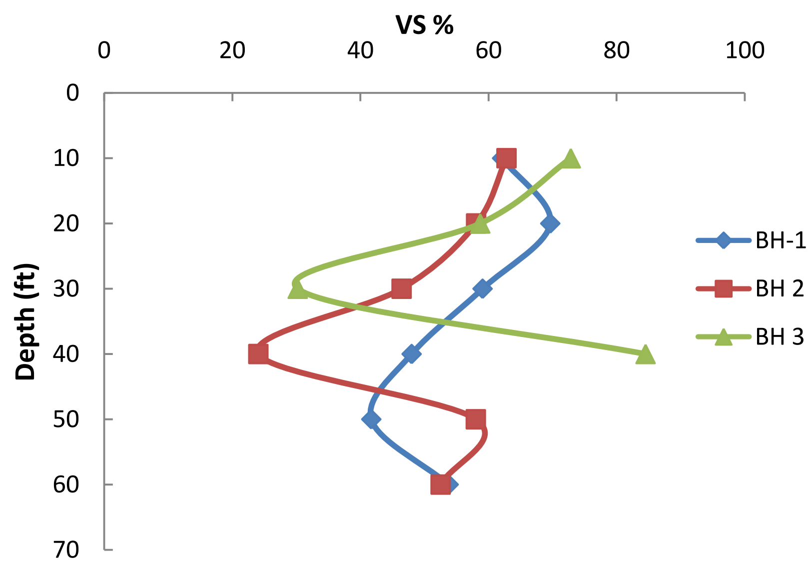


Figure 5.10 Volatile solids of samples with depth

Table 5.6 Volatile solids of samples

Boring	Depth (ft)	VS%
3D-BH-1	10	62.11
	20	69.63
	30	59.06
	40	47.98
	50	41.68
	60	53.76
3D-BH-2	10	62.79
	20	58.07
	30	46.41
	40	24.05
	50	58.01
	60	52.51
4B-BH-3	10	72.85
	20	58.71
	30	30.2
	40	84.53
Average		55.15
Standard Deviation		15.16
Maximum		84.53
Minimum		24.05

5.3 Analysis and Discussion

5.3.1 *Laboratory Test Results*

The physical composition of the collected landfilled MSW samples vary widely and no similarities in composition are observed between the MSW samples retrieved from the similar depths.

BH-1 has high fine particles and low paper and yard wastes content at 40 ft and 50 ft that indicates the higher level of degradation at these depths. Also the sample at 10 ft seems more degraded than 20 and 30 ft. Nevertheless, the composition and visual inspections show partial degradation of the samples at BH-1 with higher degraded materials at 40 ft and 50 ft.

For BH-2 at 40 ft the paper content decreases and the fine content increases. However, high paper content is observed for 50 ft and 60 ft depth. From the physical compositions, it may be assumed that for BH-2, maximum degradation occurred between 30 ft and 40 ft depth and for the rest of the MSW samples partial degradation occurred.

For BH-3 which is located in another cell, the higher degradation was observed at sample from 30 ft ; however, it is still partially degraded.

Consequently , it seems a layer at the middle level of the landfill with 30 to 40 ft depth has been experienced higher degradation; however the samples are assumed partially degraded.

The average composition has a degradable percentage of 45% (paper 27.5%, food 0.6%, textile 5% and yard waste 11.8%) which is almost half of the total composition. This indicates that the Cefe Valenzuela landfill has still substantial potential of further degradation and production of gas.

The moisture content results from the three borings BH-1, BH-2 and BH-3 shows that the average moisture content (wet weight basis) from all three boreholes are 20% ± 2%. Variations of moisture content with depth are observed; however, the moisture content of all samples ranged between 15% to 30%. At 30 ft depth, for both the borings from cell 3D (BH-1 and BH-2) and at 50 ft from BH-2 the moisture content was high (27.99%, 25.35% and 30.19% respectively on wet weight basis). This is in harmony with the conclusion of the physical composition tests about the more degraded layer at 30 ft depth. The Comparatively low moisture content results are observed from BH-3 (between 13% to 20%) which may be the reason behind the overall lower degradation in these MSW samples. The average moisture content of the landfilled MSW samples is 25.05% (wet weight basis). This value is lower than the optimum value. Therefore, leachate recirculation technique can be applied in order to increase the moisture content and enhance the biodegradation.

Regarding the unit weight, the highest unit weight for BH-1, BH-2 and BH-3 has been determined at 50 ft, 40 ft, and 30 ft respectively. This confirms the point that the middle layers are more degraded. Unit weight values have been used to predict the current porosity.

The permeability results show that the permeability of all the MSW samples ranged between 10^{-3} cm/sec to 10^{-4} cm/sec which is similar to the permeability of fresh MSW samples. This is close to the lower boundary of the unit weight range that is mentioned in Table 2.12.The maximum permeability is observed at 20 ft and 30 ft depth of BH-1, from visual observations and the composition results these two samples are very similar to fresh samples.

Based on the volatile solid test results, the MSW sample from BH-2 at 40 ft depth and from BH-3 at 30 ft depth are most degraded; however, most of the samples are partially degraded and some of the samples are almost similar to fresh MSW sample.

Consequently, the test results confirm the fresh or low degraded state of the wastes. For predicting the flow condition at current time, the high saturated hydraulic conductivity has been used.

5.3.2 *Computer Modeling Results*

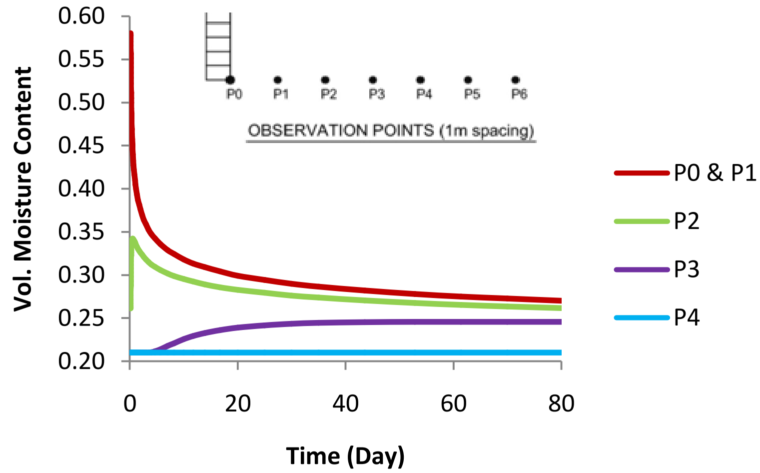
Results for single wells and well cluster are presented in this section. For single wells, results are presented for two cases of partially degraded and mostly degraded case. The current condition of landfill is similar to the partially degraded case; however, with aging and degradation the results are expected to be similar with the degraded case.

5.3.2.1 Single Well - Partially Degraded MSW

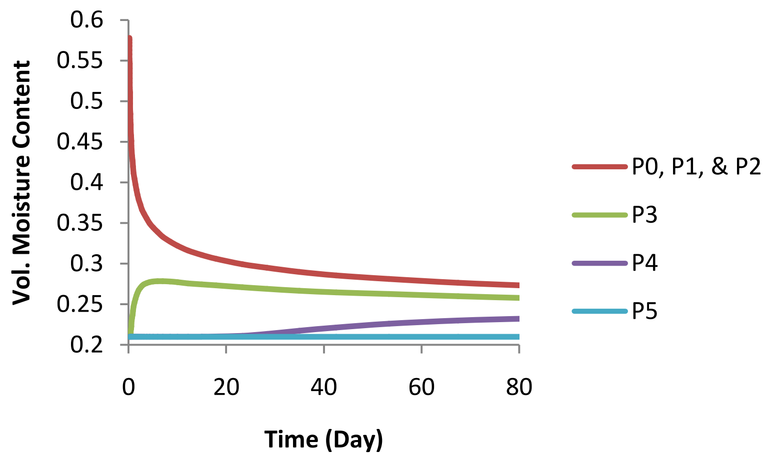
This section explains the leachate recirculation using a single well through fresh or partially degraded MSW. This case represents the high boundary of vertical hydraulic conductivity and drainage capacity. Figure 5.11 shows the volumetric water content of observation points versus time for two cases of 4000 and 8000 gallon of leachate injection.

For both cases, the moisture content at P0, P1, and P2 reaches a high value during the injection and after injection cessation it starts to decrease. One difference of two cases at Figure 5.11 is the degree of saturation of P2 just after the leachate cessation. The degree of saturation for 8000 gal case is 70% higher than the degree of saturation for 4000 gal; however, this difference decreases due to travel of excessive moisture. Moreover, the leachate distribution extent is longer for the 8000 gal case so the S at P3 is higher for this case.

Also, Figure 5.11 show that the moisture contents for the first couple of weeks go beyond the allowable range, which in this case the allowable gravimetric wet basis moisture content is 40 %. At day 80, the moisture contents are equal or below the allowable range (assuming unit weight of 50 pcf).



(a)



(b)

Figure 5.11 Volumetric moisture content vs. time at observation points after a) 4000 gallon ,and b) 8000 gallon leachate injection

Figure 5.12 shows the leachate distribution lateral extent for different leachate quantity. It demonstrates that the significant part of leachate travel is during the leachate injection. For lower quantity of 500, 1000, and 2000 gallon, the extent is almost constant after a couple of days; however, for higher quantity the excessive moisture inside the saturation bulb around the well takes time to be distributed.

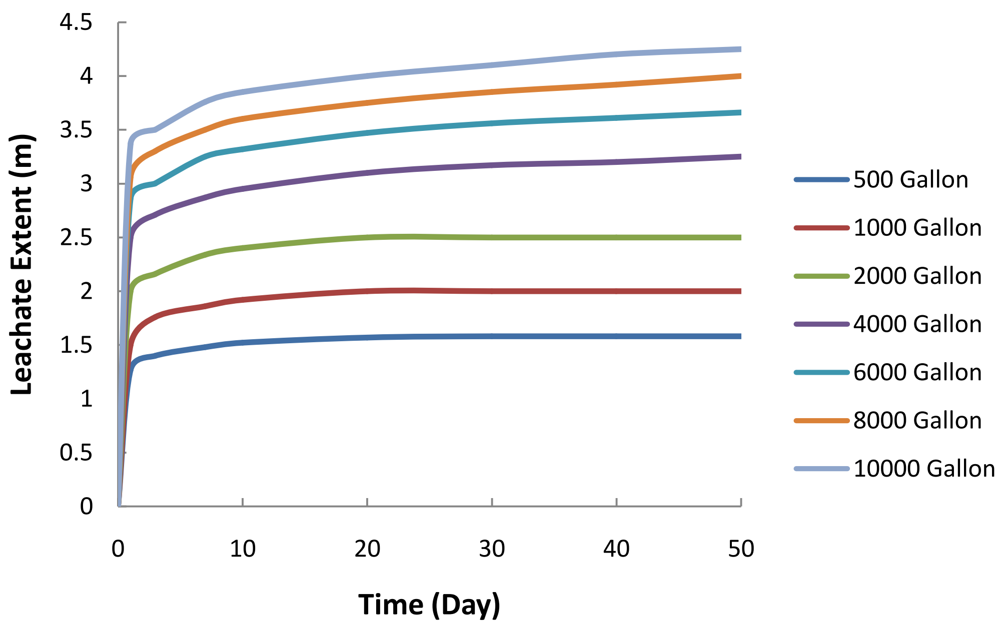
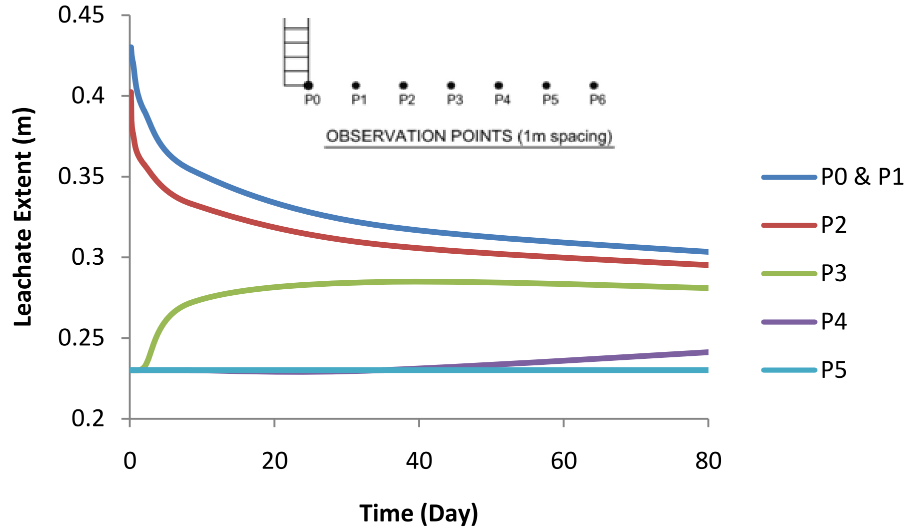


Figure 5.12 Leachate extent vs. time and leachate quantity for partially degraded MSW

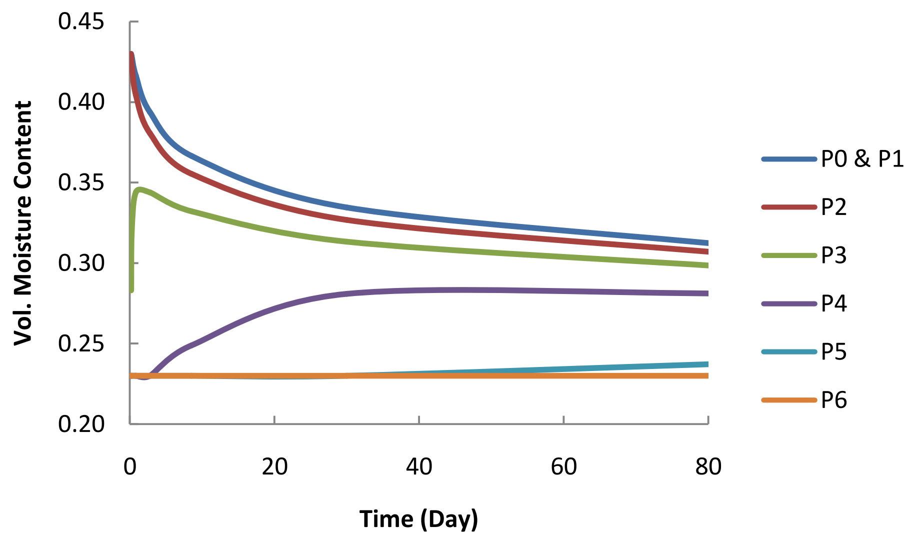
5.3.2.2 Single Well – Mostly Degraded MSW

This section explains the leachate recirculation using a single well through mostly degraded MSW. This case represents the low boundary of vertical hydraulic conductivity and drainage capacity. Therefore, higher lateral extent is anticipated.

Figure 5.13 shows the volumetric water content of observation points versus time for two cases of 4000 and 8000 gallon of leachate injection. It reflects the longer leachate extent for 8000 gallon case as P5 absorbs moisture after 40 days. Also, the moisture content for P0 to P4 is higher for 8000 gallon case; however, the difference in moisture contents decreases by time.



(a)



(b)

Figure 5.13 Volumetric moisture content vs. time at observation points after a) 4000 gallon, and b) 8000 gallon leachate injection

Figure 5.14 shows the leachate distribution lateral extent for different leachate quantity. It demonstrates that the significant part of leachate travel is during the leachate injection. The leachate distribution continues after the injection cessation because the excessive moisture inside the saturation bulb around the well takes time to be distributed.

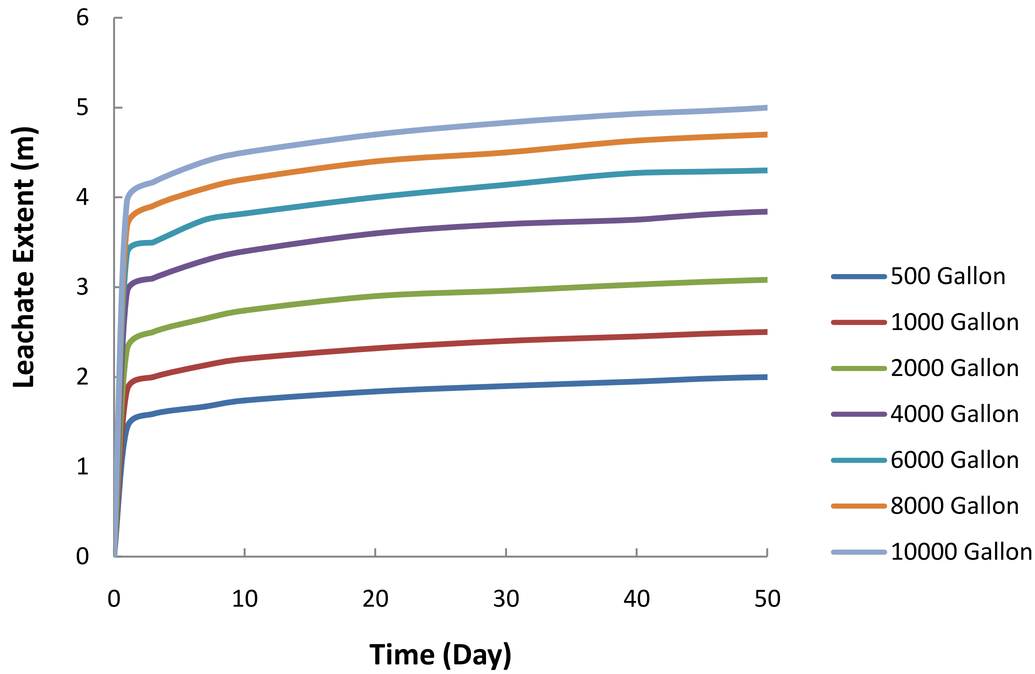
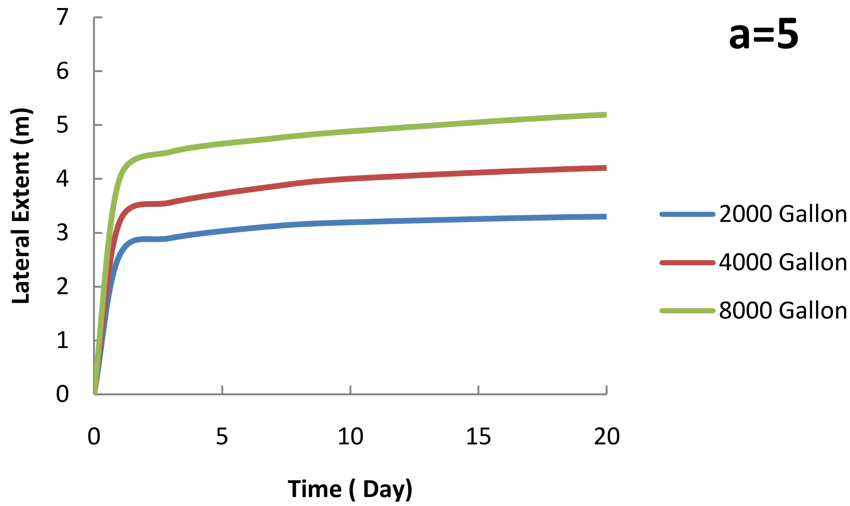


Figure 5.14 Leachate extent vs. time and leachate quantity for mostly degraded MSW

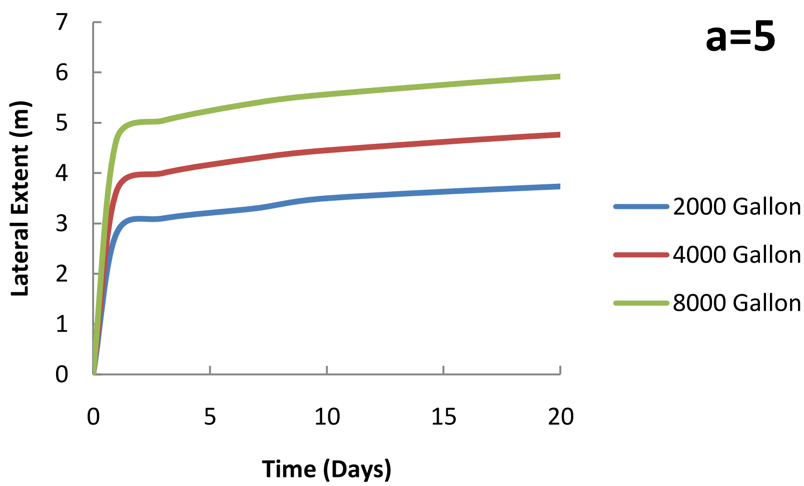
5.3.2.3 Single Well – Anisotropic Condition

Although the anisotropy factor of MSW has not been measures at field, the modeling results for the City of Denton Landfill showed the anisotropic characteristics of the MSW. Moreover, according to Townsend (1995) and Landva et al. (1998) landfill MSW is strongly anticipated as an isotropic media due to the compaction practice at field. Also, the difference of vertical and horizontal stresses in addition to the anisotropic deposition of particles results in anisotropy. Anisotropic factor ($a= k_r/k_z$) more than 1 increases the leachate distribution extent significantly.

In this section the simulation results under two anisotropic condittions, a=5 and a=10, have been represented. For this study, the in-situ anisotropic factor is not measured, so the maximum value has been used to develop the charts. The anisotropy factor may reach 10 (Landva et al., 1998). The lateral extents for two anisotropy factors of a=5 and a=10 are shown in Figure 5.15 and Figure 5.16, respectively.

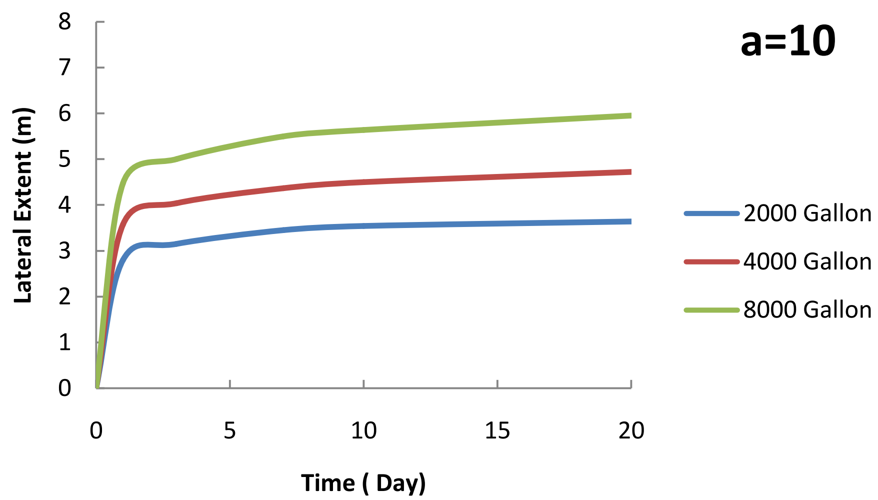


(a)

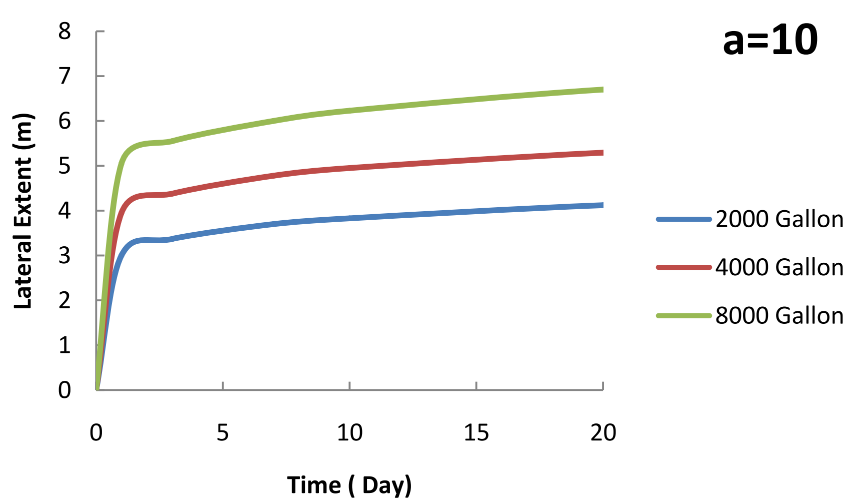


(b)

Figure 5.15 Leachate extent vs. time and leachate quantity for anisotropic a) partially degraded MSW, and b) mostly degraded MSW

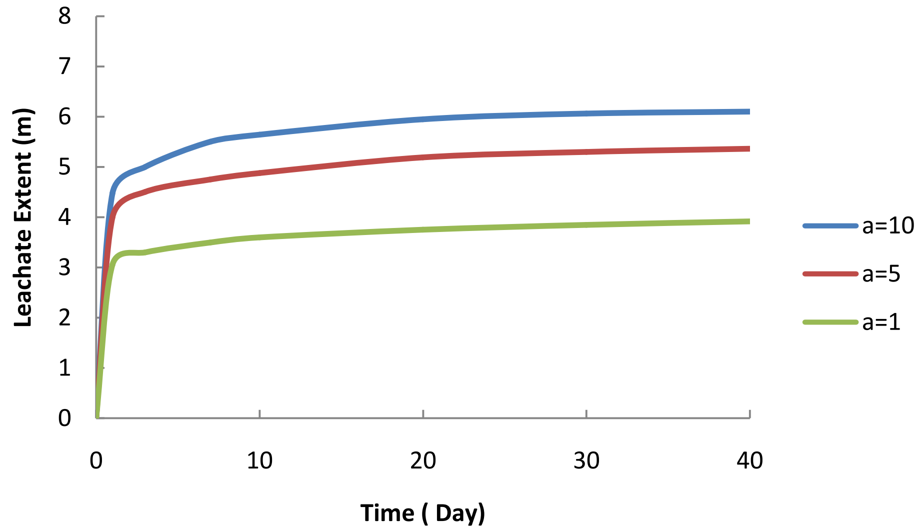


(a)

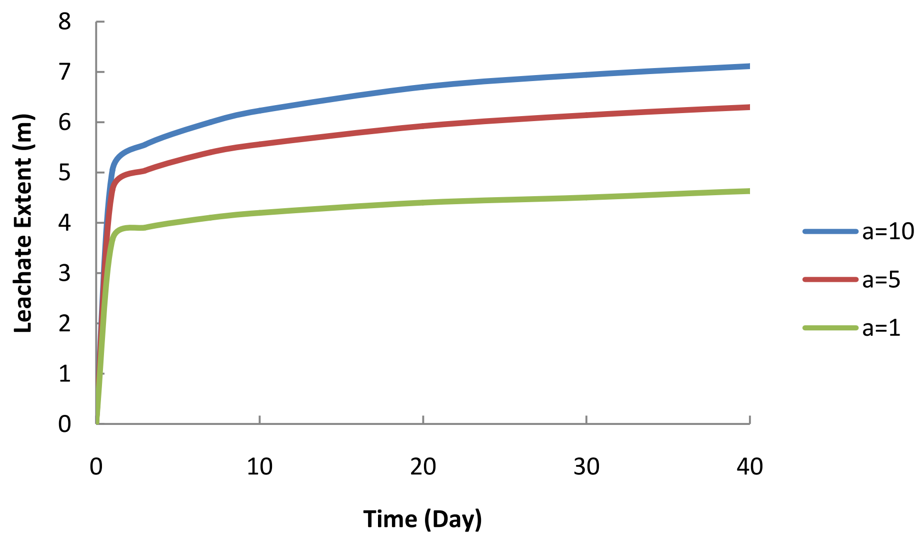


(b)

Figure 5.16 Leachate extent vs. time and leachate quantity for anisotropic a) partially degraded MSW, and b) mostly degraded MSW



(a)



(b)

Figure 5.17 Effect of anisotropy factor on leachate extent after 8000 gallon leachate injection in a) partially degraded MSW, and b) Mostly degraded MSW

Figure 5.17 show the effect of anisotropy on leachate extent. In Figure 5.17 (a), at day 40 the leachate extent for a=5 and a=10 is 37% and 56% higher than the isotropic condition, respectively. In Figure 5.17 (b), at day 40 the leachate extent for a=5 and a=10 is 38% and 55% higher than the isotropic condition, respectively. Therefore, anisotropy in addition to the saturated hydraulic conductivity is one of the most important parameters that determine the leach ate extent.

5.3.2.4 Single Well – Effect of Degradation on Leachate Extent

Degradation state of the MSW is one of the affecting parameter on leachate lateral extent. Indeed, the more degraded MSW has less permeability. This reduces the gravity drainage capacity. Therefore the affected area as shown in Figure 5.19 becomes more circular rather than the oval shape, which is the affected area for partially degraded MSW. All the simulations, including the anisotropic conditions, confirm that in more degraded MSW leachate travels more distant laterally.

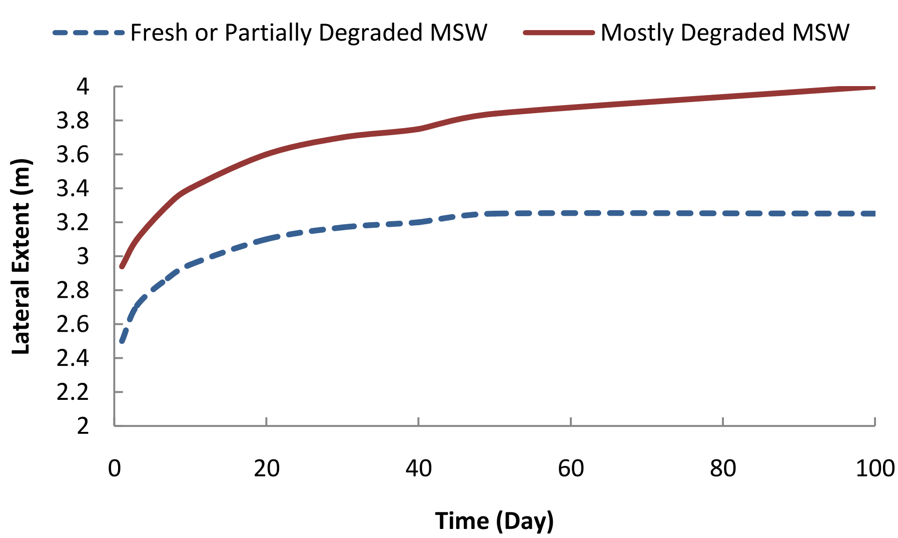
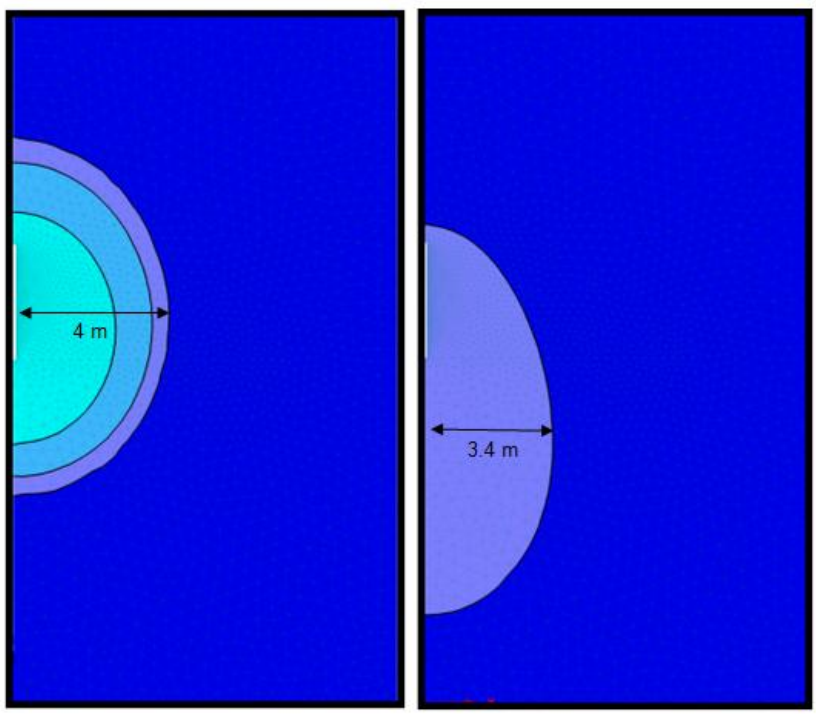


Figure 5.18 Effect of degradation on leachate lateral extent (4000 gallon)

Figure 5.18 shows the effect of degradation on leachate extent for an exemplary case. Although the leachate quantity is same for both case, lateral extent for the mostly degraded MSW is always higher than the partially degraded.

Figure 5.18 shows another effect of degradation which is the moisture profile stabilization. For the fresh MSW, after 40 days the lateral extent is almost constant; however, for the mostly degraded MSW still is going on after 40 days. This is because the higher capacity of finer particles in the mostly degraded MSW to trap the moisture. However, the excessive moisture will be distributed in a longer time, which here is almost 100 days.



(a) (b)

Figure 5.19 Affected zone after 100 days for 4000 leachate injection in a) mostly degraded MSW, and b) fresh or partially degraded MSW

Consequently, the degradation state of MSW significantly alters the saturation profile around the well. It changes the unsaturated flow parameters as well as the saturated hydraulic conductivity. As the results in this study are based on two boundaries of degradation, the actual condition may place among these two conditions. Therefore, interpolation may be required to obtain a more realistic value for a specific condition. Figure 5.20 shows the saturation profile around a recirculation well after 100 days for two different leachate quantity and degradation state. The wetted areas have been demonstrated in Figure 5.21. Although the lateral extent for different degradation state differs, Figure 5.21 shows that the wetted area is almost same for a same leachate quantity. Also, the saturation profiles for the mostly degraded case contain larger area with higher volumetric water content because of the lower porosity in mostly degraded MSW.

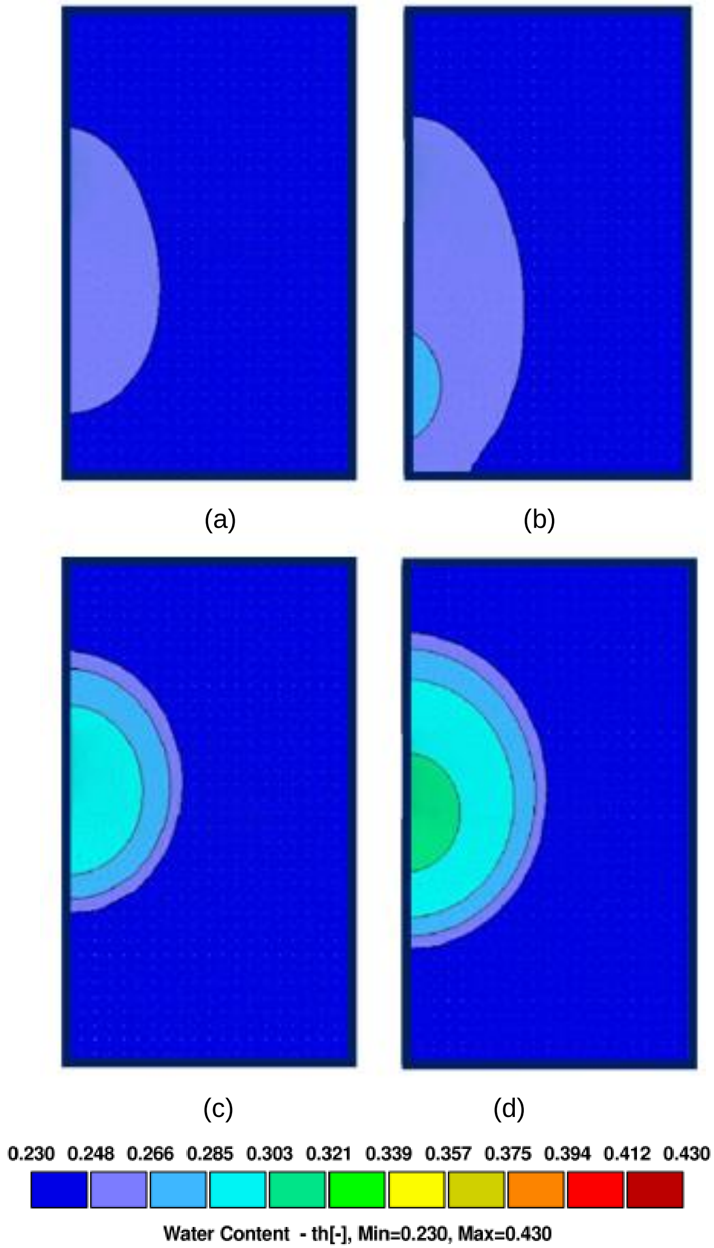


Figure 5.20 Saturation profile around the well for injection of a) 4000 gal in fresh MSW, b) 8000 gal in fresh MSW, c) 4000 gal in mostly degraded MSW, and d) 8000 gal in mostly degraded MSW

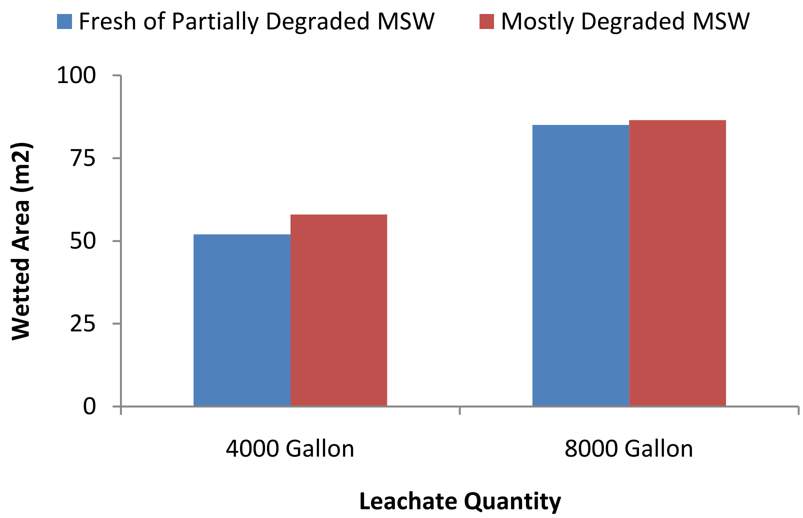


Figure 5.21 Wetted area vs. leachate quantity and degradation state after 100 days

5.3.2.5 Well Cluster

Well cluster is a group of vertical wells that are located close together. Well clusters based on their arrangement can affect larger area. In this section the efficiency of well clusters will be discussed. All simulations in this section are based on the fresh or partially degraded MSW properties. This represents the condition more similar to the current field condition. Also, the simulation results for the City of Denton Landfill confirmed that the partially degraded MSW properties are appropriately reflecting the field condition. Moreover, as it was discussed earlier, leachate travels more distance laterally in the mostly degraded MSW. Therefore, the results in this section show the lower boundary of the lateral extent which is the more conservative approach for design purposes.

Figure 5.22 shows the behavior of well cluster for three arrangements of wells. One arrangement includes two wells while another one includes three wells. The geometric details are shown in Figure 4.7. As it is shown in Figure 5.22 from top to bottom, the cluster that includes more wells can affect larger area.

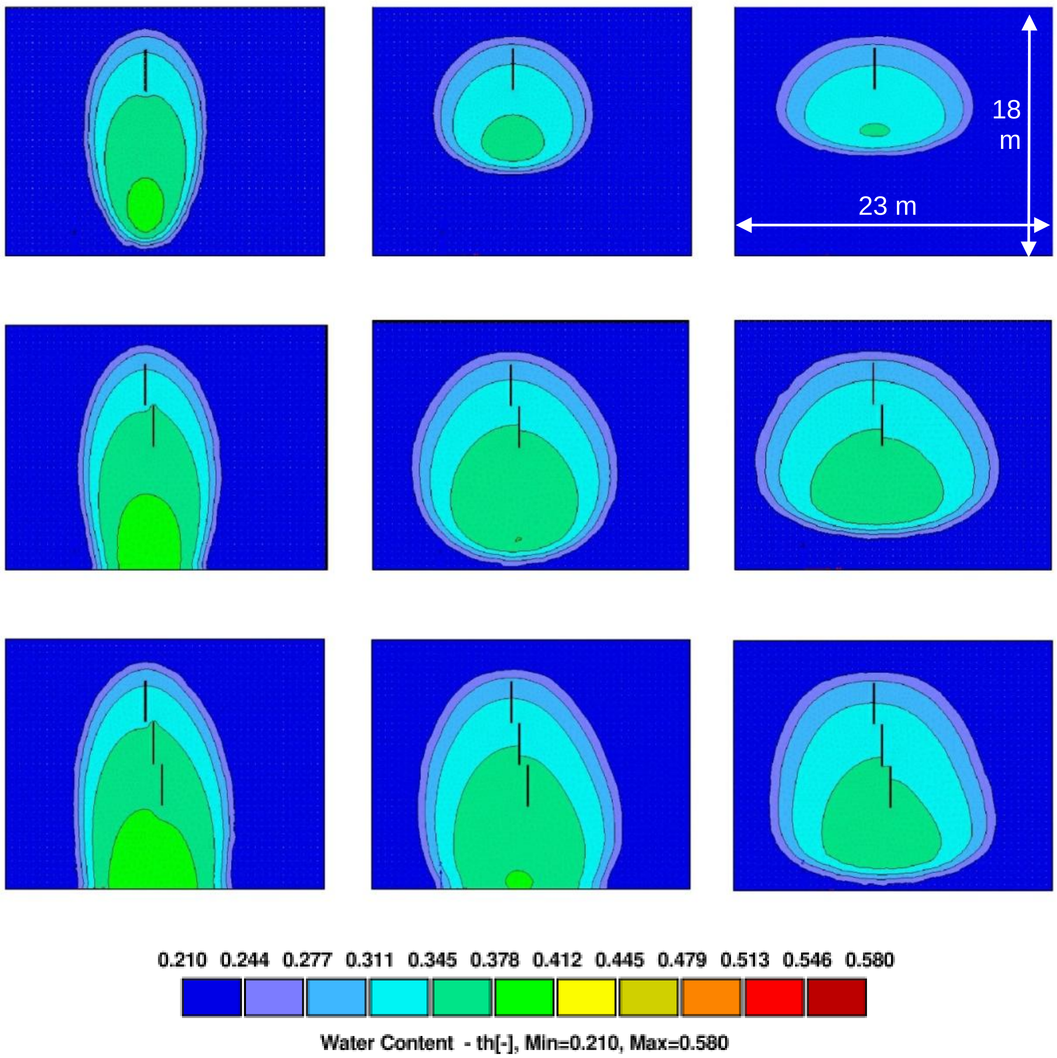


Figure 5.22 Saturation profile 7 days after 20,000 gallon leachate injection per well for three arrangements of vertical wells. From left to right the anisotropy factor is 1, 5, 10.

Although the number of wells is an important parameter to cover larger area, more wells necessarily do not result the best design. Indeed, the position of wells is an important parameter beside their number. As it is shown in Figure 5.23, the wetter area for the cluster of two wells is significantly greater than that of single well. ‘The wetted area in the cluster of two wells is 100% greater than the single well. On the other hand, the cluster of three wells shows no significant affect in comparison with the cluster of two wells. It just shows 7 % increase in the affected area. The inefficiency of the cluster of three wells is caused by the short circuit of leachate between the lowest well and the landfill drainage system. Indeed, the vertical distance between the bottom level of the lowest well and the drainage system is not long enough. Therefore, the leachate reaches the drainage system before covering larger area.

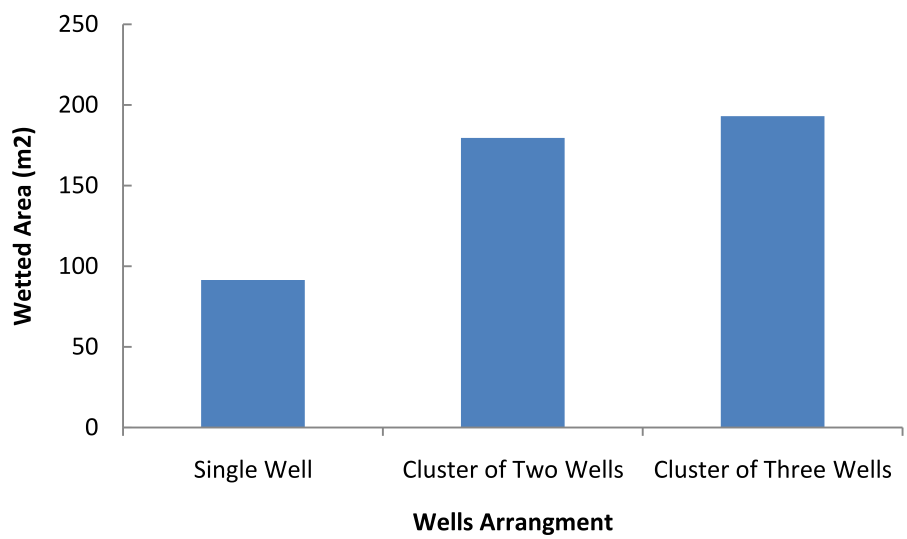


Figure 5.23 Wetted area for three cluster arrangements (Based on the results for anisotropy factor of 5 in Figure 5.22)

Consequently, the position of wells is as important as their number. The simulation in this section showed that the cluster of three wells is not an efficient arrangement for the vertical wells unless the location of the lowest well varies.

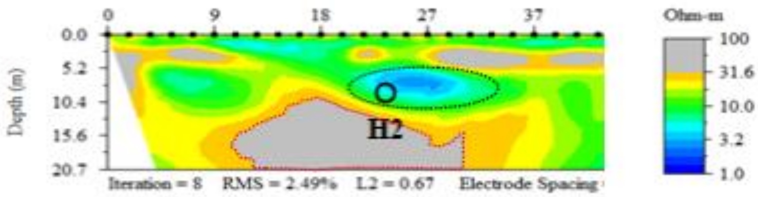
5.3.3 *Model Validation*

Although the simulation results in this study have not been confirmed by the field data yet, the assumptions and model input data have been validated, using field data from the City of Denton Landfill. The City of Denton Landfill is an Enhanced Leachate Recirculation (ELR) landfill that uses horizontal trenches and permeable blankets to recirculate the leachate.

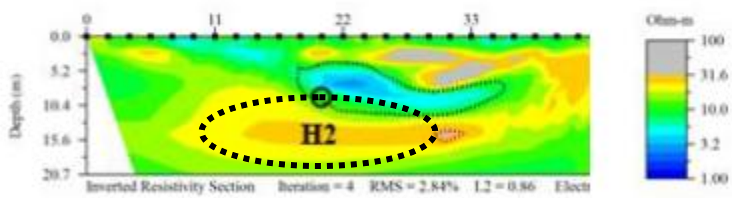
According to Samir (2011), MSW at the cell which is the area of interest is fresh or partially degraded in most locations. Therefore, partially degraded material properties have been assumed as the MSW flow properties. Leachate quantity of 2500 gallon was used for this simulation. This value was obtained by back calculation. The RI test results are based on 5500 gallon leachate injection through the total length of pipe; however, using this quantity is not helpful because the leachate in not distributed uniformly along the horizontal trench.

To simulate the wetted area, which is shown in Figure 5.24, the important parameters including the waste hydraulic conductivity, waste porosity, and initial moisture content were manipulated. No parameter found effective but the anisotropy factor. Indeed for the studied case the horizontal permeability was significantly higher than the vertical permeability.

Also, the fresh or partially degraded material properties reflected the MSW flow properties correctly. As it is shown in Figure 5.24, the horizontal trench is located at the top of the oval wetted area. Simulations with the degraded MSW properties show the trench at the center of the oval wetted area. Indeed, the leachate in the degraded case significantly goes upward, which is not the case here. It should be notices that the blue area above the pipe has higher initial moisture content before injection. Therefore, because of its higher hydraulic conductivity a portion of leachate flows through it. Figure 5.25 shows another RI test results which has the same features in saturation profile with the pipe H2.



(a)



(b)

Figure 5.24 RI test results for a recirculation pipe H2 a) before injection, and b) 7 days after injection

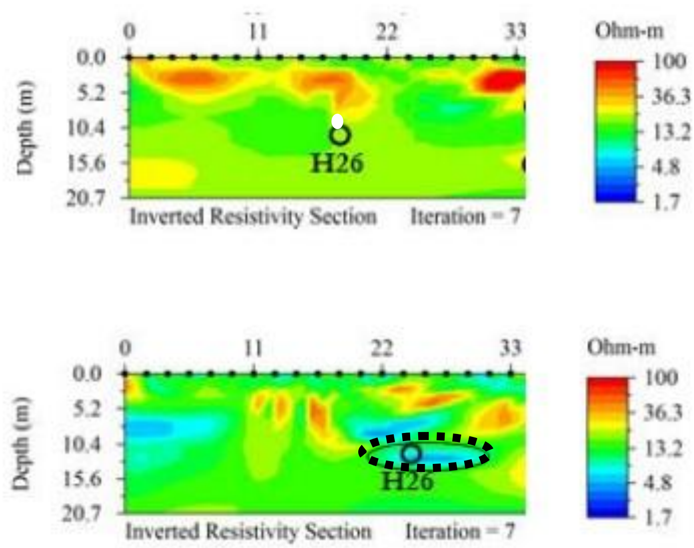


Figure 5.25 test results for a recirculation pipe H26

Figure 5.26 shows the simulation results which reflect the RI test results correctly. The wetted areas at day 7 for simulation and RI results are matching. To make such an oval shape the anisotropy factor of 10 was used in simulation. Using this anisotropy factor reaches the extent to almost 10 m each side, which is the extent at field.

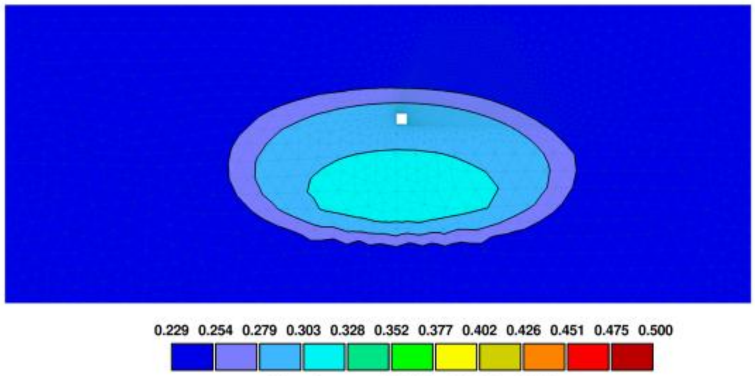


Figure 5.26 Simulation of pipe H2

Although the simulation based on fresh or the partially degraded MSW properties are matching with the RI test results in Figure 5.25 and Figure 5.26, there are some other RI test results that are not matching with the simulations. Compacted daily covers, various initial water content, physical composition variation, local degradation variation are some of the affecting parameters that may change the leachate distribution pattern.

As the Cefe Valenzuela landfill is located in a same geographical region and the waste stream is similar to the City of Denton Landfill, the results based on the fresh or partially degraded properties are suggested to be considered for the current condition.

Chapter 6

Summary and Conclusions

The main objective of this study was to study the characteristics of MSW samples collected from Corpus Christi landfill and to model leachate recirculation using vertical wells in bioreactor landfill operations. The purpose of leachate recirculation is to increase the moisture content up to the optimum value in order to maximize the biodegradation rate. Prediction of leachate distribution after the injection is important to enhance the performance of bioreactor landfills. By knowing the leachate distribution extents, the leachate quantity or the wells arrangement can be determined in order to maximize the leachate recirculation effects.

The Cefe Valenzuela landfill was the selected study area. This is a conventional landfill which already has received a permit to operate as a bioreactor landfill. To characterize the MSW, an experimental program was performed. Laboratory tests showed that most of the collected samples are in a fresh or low degraded condition. This fact helps to predict the leachate distribution pattern for the current time.

Simulations using HYDRUS-2D showed the effect of different parameters such as leachate quantity, MSW flow properties, and degradation state and anisotropy factor on the leachate distribution.

6.1 Summary and Conclusions

The work completed for the present study can be summarized as follows:

1. For this study samples are collected from three boreholes. Total of 16 samples from different depth, 10 ft to 60 ft, were collected. Each sample weighed approximately 15 to 20 Lbs.
2. An extensive experimental program was performed to characterize the landfilled MSW samples. Test result showed the fresh or low degraded condition for most of the collected samples. This is an important fact to predict the current flow properties of the landfill.
3. Conceptual model was developed based on the actual field condition and the computer model HYDRUS-2D features. Models were developed for a single well and well cluster. MSW flow properties were assumed based on the previous study results and the model validation. Two cases of MSW flow properties were assumed to represent the possible properties range at the field. One case is based on the fresh or low degraded MSW condition while the other is based on the mostly degraded condition. Finally, the methodology was validated based on the City of Denton landfill leachate recirculation experience.
4. Leachate recirculation using a single vertical well was simulated for different conditions. The well was far from the bottom drainage system that all the injected leachate was absorbed by MSW. The leachate lateral extent is higher in the mostly degraded MSW; however, the area that injected leachate covers in both cases is mainly dependant on the leachate quantity.
5. Leachate recirculation through a well cluster was simulated to evaluate the efficiency of well clusters. Cluster of two wells and cluster of three wells was considered as the arrangements. Cluster of two wells was an efficient cluster

because it doubled the wetted area. However, the cluster of three wells did not increase the wet area in comparison with the cluster of two wells. The main reason was short distance between the lowest well and the drainage system which was 6 m (20 ft).

6. In the mostly degraded MSW leachate distribution zone is more circular shape, which the well is located at the center. In fresh or partially degraded MSW, the zone is more oval shape with the longer vertical axis and the well is located at the top. Indeed, in the mostly degraded MSW a significant portion of leachate travels upward. Therefore, the placement of well close to the landfill top cover system may create seepage hazards.
7. The conceptual model was validated using the City of Denton landfill. Although the recirculation technique was different in this landfill, there are many similarities between the two landfills. The degradation state and the physical composition of two landfills are similar. Therefore, same MSW flow properties as the Cefe Valenzuela landfill were used. The simulation results were matching with most of the RI test results. However, there were some other RI test results that because were not matching. Compacted daily covers, various initial water content, physical composition variation, local degradation variation may cause their pattern. It is clear that MSW is so heterogeneous and complex in its structure. Therefore always there is possibility that the leachate does not follow the predicted pattern.

6.2 Recommendations for Future Studies

To enhance the reliability of the simulations and to make the current study even more effective, it is recommended that the work is further continued as mentioned in this section:

1. Unsaturated flow properties of MSW can be determined using the laboratory methodologies.
2. In-situ tests can be performed to determine the anisotropy factor of MSW.
3. The model can be calibrated using the field data. The RI tests can be performed to monitor the leachate distribution.

Appendix A Saturation Profiles

Volumetric Water Content Profiles after 8000 Gallon Leachate Injection

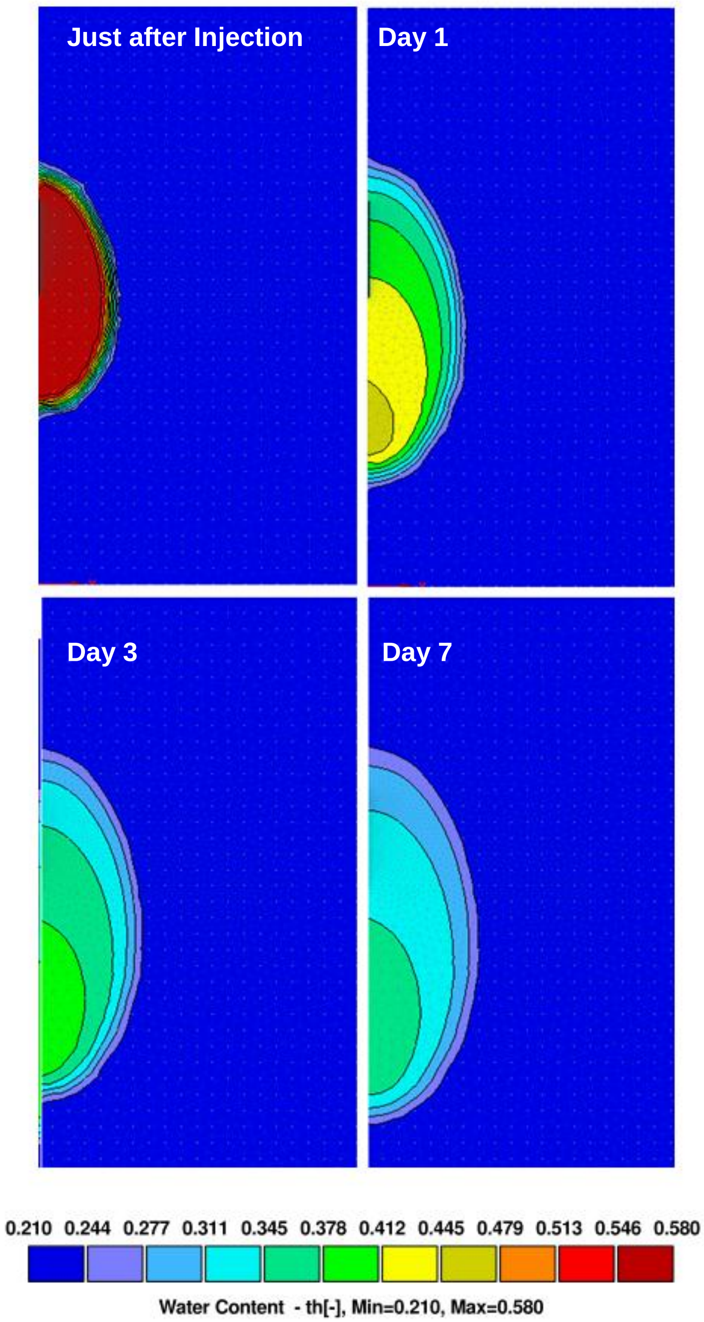


Figure A.1 Isotropic fresh or partially degraded MSW

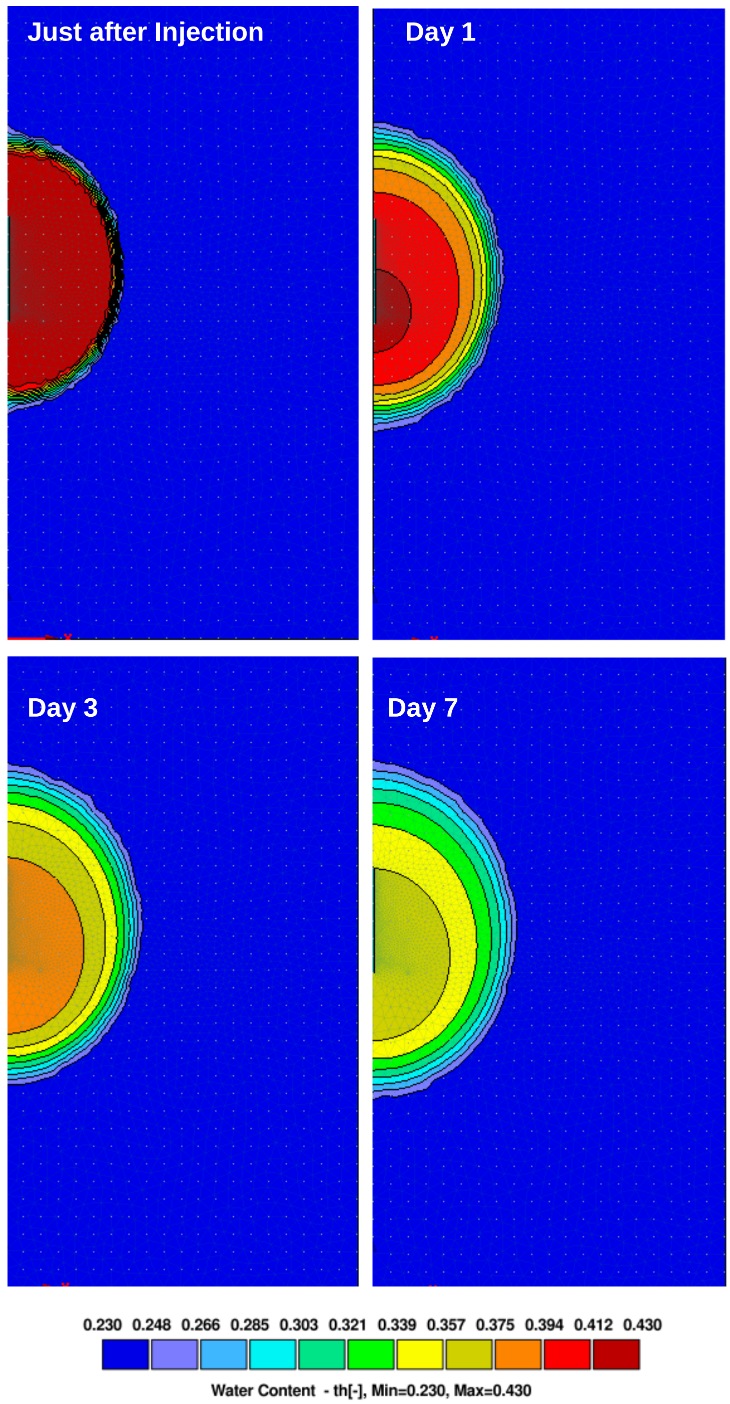


Figure A.2 Isotropic mostly degraded MSW

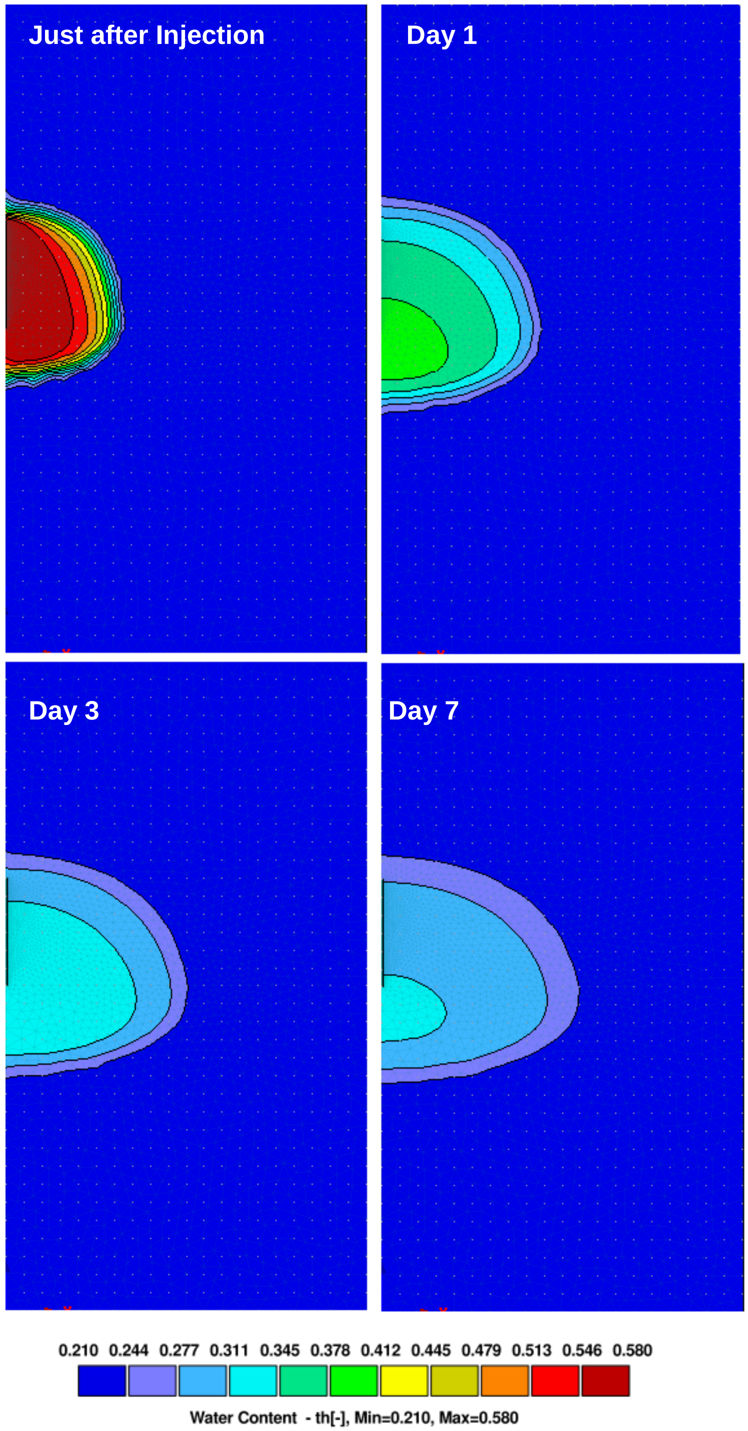


Figure A. 3 Anisotropic (a=10) fresh or partially degraded MSW

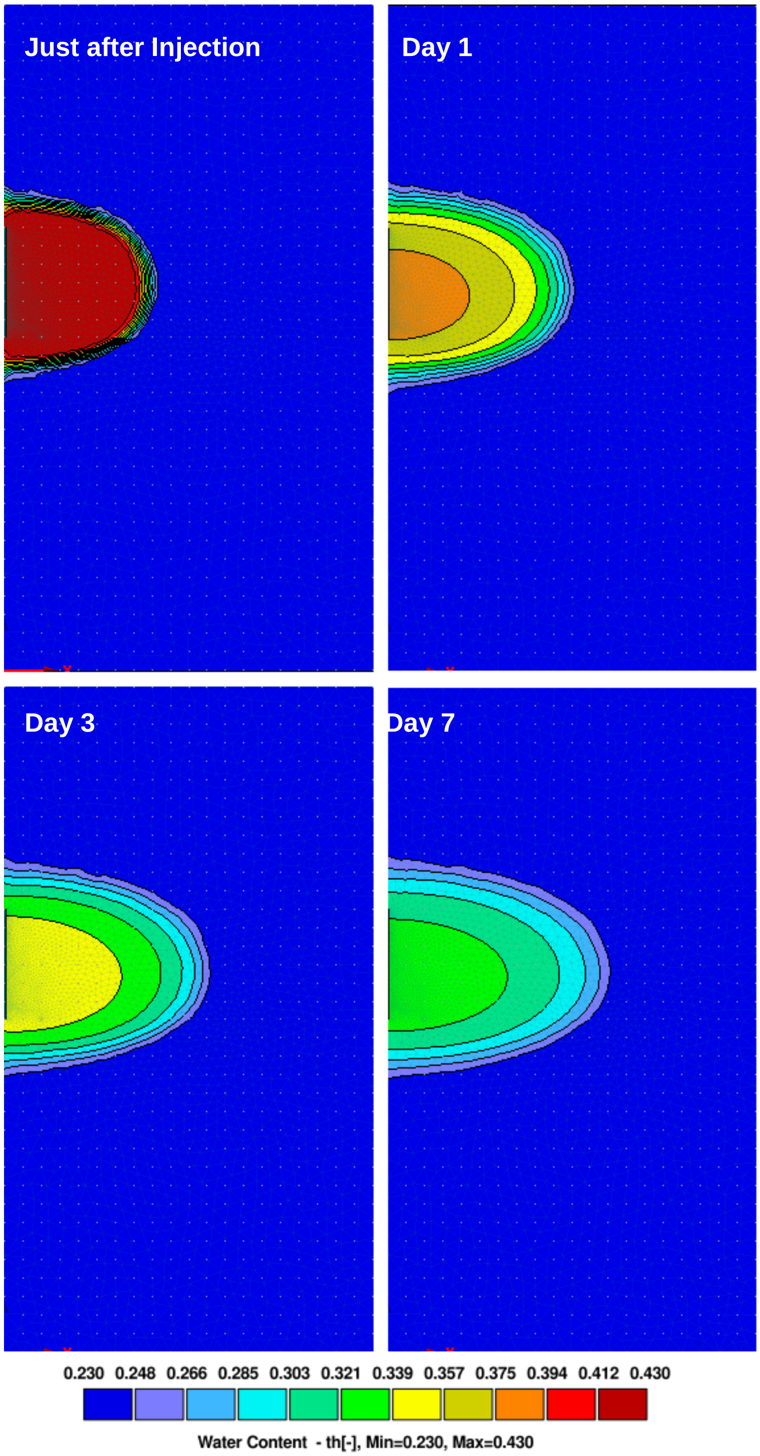


Figure A.4 Anisotropic (a=10) mostly degraded MSW

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