Two-Phase Modeling of Leachate Recirculation Using Vertical Wells in Bioreactor Landfills

Krishna R. Reddy, F.ASCE; Hanumanth S. Kulkarni; and Milind V. Khire, M.ASCE

Abstract: Leachate recirculation or liquid injection is an established practice for operating landfills as bioreactors to enhance the biodegradation of municipal solid waste (MSW). Among other subsurface liquid injection methods, vertical wells (VWs) represent one of the most common methods used for active or closed landfills. The current design and operation of VW systems does not consider the effect of additional liquid injection on the increase in gas pressures. In this study, a two-phase model that assumes landfill leachate and gas as immiscible phases was used to predict the moisture distribution and pore water and pore-gas pressures in a typical bioreactor landfill that uses VWs as its leachate injection or recirculation system. The unsaturated liquid and gas properties of MSW were simulated based on the van Genuchten model. The study evaluates the effect of the unsaturated hydraulic conductivity of MSW, the heterogeneous and anisotropic nature of the MSW, and the geometric configuration of VWs on moisture distribution and pore water and gas pressures. The unsaturated hydraulic properties of MSW significantly influence the wetted area and pore-water pressures during the initial stages of leachate injection and during the gravity drainage that follows. The numerical modeling results show that the gas pressures in the landfill will be of the order of 60 to 150 kPa, which are far above the liquid pressures for most typical liquid injection rates when no gas accumulation in the landfill is assumed due to an active gas extraction system. Hence, the slope stability of bioreactor landfills needs to be assessed by using the effects of gas and liquid pressures, especially when the gas extraction system is operating at below optimum efficiency.

CE Database subject headings: Landfills; Leaching; Municipal wastes; Solid wastes; Unsaturated flow; Moisture; Pore water; Pore pressure.

Author keywords: Bioreactor landfills; Leachate recirculation; Vertical wells; Municipal solid waste; Unsaturated flow; Moisture distribution; Pore-water pressure; Pore-gas pressure.

Introduction

The disposal of municipal solid waste (MSW) in engineered landfills will continue for foreseeable future in the United States and other countries. The modern landfills, because of relatively impermeable liners and efficient leachate and gas collection systems, cause MSW to exist under relatively dry conditions; this is not conducive to a faster biodegradation of the organic fraction of MSW (Reinhart and Townsend 1997; Sharma and Reddy 2004). Bioreactor landfills involving the recirculation of leachate through the landfill are being considered to successfully increase the moisture and enhance the distribution of nutrients and microbes to accelerate the biodegradation of the MSW [Reinhart and Townsend 1997; Reddy 2006; Interstate Technology and Regulatory Council (ITRC) 2006].

Various field systems are available for the recirculation of leachate in landfills, including vertical wells (VWs) (Khire and Mukherjee 2007; Reddy 2006; Jain et al. 2010), horizontal trenches (HTs) (McCreanor 1998; Haydar and Khire 2005), and drainage blankets (DBs) (Khire and Haydar 2007; Haydar and Khire 2007). A typical VW consists of a perforated pipe surrounded by highly permeable filter material (usually gravel) located within the desired depth interval of a vertical well. The key advantage of a VW is that one can be installed in active or closed landfills. The design of VWs should be such that uniform and adequate moisture distribution is achieved throughout the landfill without inducing excessive pore pressures, which endanger the stability of the landfill and may result in excessive differential settlements. Currently, the design of VWs ignores the effect of increase in gas pressures as a result of the additional liquid pressures used to inject the liquid. Mukherjee and Khire (2012) have shown, using controlled large-scale liquid injection experiments, that gas (air) pressures can exceed the liquid pressures during liquid injection. Hence, it is critical to understand the effects of a vertical well system on the moisture distribution and pressures of pore liquid and coupled gas within a landfill and, ultimately, to develop a rational methodology for the design of VWs.

A numerical two-phase flow model was used in this study to evaluate the effects of the unsaturated hydraulic conductivity of MSW, the heterogeneous and anisotropic nature of MSW, and the geometric configurations of VW systems on moisture distribution and pore water and pore-gas pressures.

Numerical Two-Phase Flow Model

The two-phase model assumes the flow of two immiscible fluids (leachate as wetting fluid and landfill gas as nonwetting fluid) to fill
the pore spaces of the MSW. The flow of each fluid is described by Darcy’s law, with the unsaturated hydraulic conductivity represented by a van Genuchten function [Peaceman 1977; Lu and Likos 2004; Itasca Consulting Group Inc. (IGCI) 2011]. According to Darcy’s law, the transport of wetting leachate (with superscript \( L \)) and nonwetting landfill gas (with superscript \( G \)) fluids is given by

\[
q^L_i = -k^L_{ij} k^L \frac{\partial}{\partial x_j} (P_L - \rho_L g \xi^L_k) \tag{1}
\]

\[
q^G_i = -k^G_{ij} k^G \frac{\partial}{\partial x_j} (P_G - \rho_G g \xi^L_k) \tag{2}
\]

where \( k^L_{ij} \) = saturated mobility coefficient (tensor) defined as ratio of intrinsic permeability to dynamic viscosity; \( i \) = number of zones in horizontal (\( x \)) direction; \( j \) = number of zones in vertical (\( y \)) direction; \( \kappa_i \) = relative permeability for the fluid (function of saturation); \( \mu \) = dynamic viscosity; \( P \) = pore pressure; \( \rho \) = fluid density; and \( g \) = gravity.

The relative permeabilities of the fluids are related to wetting fluid saturation (\( S_w \)) and are expressed by the van Genuchten function as (Mualem 1976; van Genuchten 1980):

\[
k^L_{eff} = S_w^b [1 + (1 - S_w^{1/a})^{1/b}] \tag{3}
\]

\[
k^G_{eff} = (1 - S_w)^c (1 - S_w^{1/a})^{2a} \tag{4}
\]

\[
S_e = \frac{S_L - S_L^b}{1 - S_L^b} \tag{5}
\]

where \( a \), \( b \), and \( \alpha \) are constants; \( S_e \) = effective saturation; \( S_L^b \) = residual wetting fluid (leachate) saturation. The van Genuchten function is used to relate \( P_e \) and \( S_e \) by

\[
P_e = \frac{\rho_L g}{\alpha S_e^{1/a}} (S_e^{1/a} - 1) \tag{6}
\]

where \( P_e \) = capillary pressure; \( \rho_L \) = wetting fluid density; \( g \) = gravity; \( \alpha \) = coefficient related to the intrinsic permeability (\( \kappa \)) and is given by

\[
\alpha = \frac{\rho_L g \sqrt{\kappa / n}}{\sigma} \tag{7}
\]

where \( n \) = porosity; and \( \sigma \) = surface tension of matrix. The sum of the saturation of wetting fluid (\( S_L \)) and nonwetting fluid (\( S_G \)) should be

\[
S_L + S_G = 1 \tag{8}
\]

Moreover, capillary pressure is related to the pressure difference between the wetting and nonwetting fluids as

\[
P_G - P_L = P_e (S_L) \tag{9}
\]

where \( P_G \) = pressure created by nonwetting fluid; \( P_L \) = pressure created by wetting fluid; \( P_e (S_L) \) = capillary pressure, which is a function of degree of wetting fluid saturation (\( S_L \)).

Fluid balance laws are available for the slightly compressible fluids and provide the variation of fluid content (variation of fluid volume per unit volume of porous material) with respect to the intensity of the volumetric fluid source. The balance laws for wetting and nonwetting fluids are

\[
\frac{\partial S^L_t}{\partial t} = -\frac{\partial q^L_i}{\partial x_i} + q^L_e \tag{10a}
\]

\[
\frac{\partial S^G_t}{\partial t} = -\frac{\partial q^G_i}{\partial x_i} + q^G_e \tag{10b}
\]

where \( \xi \) = variation of fluid volume per unit volume of porous material; and \( q_e \) = volumetric fluid source intensity.

Constitutive laws for fluids are solved for the pressures in wetting and nonwetting fluids, and saturation in wetting and nonwetting fluids:

\[
S_L \frac{\partial P_L}{\partial t} = \frac{K_L}{n} \left( \frac{\partial S^L}{\partial t} - n \frac{\partial S_L}{\partial t} - S_L \frac{\partial \varepsilon}{\partial t} \right) \tag{11a}
\]

\[
S_G \frac{\partial P_G}{\partial t} = \frac{K_G}{n} \left( \frac{\partial S^G}{\partial t} - n \frac{\partial S_G}{\partial t} - S_G \frac{\partial \varepsilon}{\partial t} \right) \tag{11b}
\]

where \( \varepsilon \) = volumetric strain; \( K_G \) = bulk modulus of gas; and \( K_L \) = bulk modulus of liquid. By combining these equations with fluid balance laws:

\[
n \left( \frac{S_L \frac{\partial P_L}{\partial t} + \frac{\partial S_L}{\partial t}}{K_L} \right) = -\left( \frac{\partial q^L_i}{\partial x_i} + S_L \frac{\partial \varepsilon}{\partial t} \right) \tag{12a}
\]

\[
n \left( \frac{S_G \frac{\partial P_G}{\partial t} + \frac{\partial S_G}{\partial t}}{K_G} \right) = -\left( \frac{\partial q^G_i}{\partial x_i} + S_G \frac{\partial \varepsilon}{\partial t} \right) \tag{12b}
\]

This produces a nonlinear system of four coupled fluid flow mechanical equations that will be solved for four unknowns: \( P_L, P_G, S_L, \) and \( S_G \). The formulation considers the mechanical compression through the term \( \varepsilon \); however, in this study, mechanical compression is neglected and only fluid flow is considered; therefore, the term \( \partial \varepsilon / \partial t \) is omitted.

Governing Eqs. (12a) and (12b) of two-phase unsaturated flow are solved numerically with the Fast Lagrangian Analysis of Continua (FLAC) program by using the finite difference method (IGCI 2011). The use of the FLAC model is validated by reported landfill related laboratory and field studies and by previous modeling studies by Kulkarni (2012). The FLAC model can predict the laboratory, field, and modeling results reasonably well (Kulkarni 2012). Landfill gas generation and landfill gas collection systems were not considered in this modeling study.

**Model Implementation**

**Conceptual Model**

A two-dimensional (2D) model of a bioreactor landfill cell, 100 m wide by 20 m high, was created in FLAC by using a graphical interface to investigate the effects of unsaturated hydraulic properties, heterogeneous MSW and mode of leachate injection. This model domain and overall modeling approach is similar to that reported by Khire and Mukherjee (2007), who used the single-phase model HYDRUS-2D (Simunek et al. 1999).

This study focuses only on leachate injection and flow through MSW when the landfill is active. Hence, the model does not include the effect of a landfill cover system. AVV having a 0.3 m diameter and screen height (\( H_s \)) of 3 m is assumed to exist at the center of the model domain (Fig. 1). The depth of the VW from the ground surface (\( H_{v} \)) and the distance between the bottom of the VW and the leachate collection system (\( d_{w} \)) were selected based on the study reported by Khire and Mukherjee (2007). A low permeability bentonite clay (with \( k_s = 10^{-7} \text{ cm/s} \)) seal is assumed above the well screen. Thus, the injection of the leachate can only occur.
through the screen depth interval. For the investigation of the effects of unsaturated hydraulic conductivity, the values of $H_w$ and $d_v$ are 11.5 and 8.5 m, respectively (Fig. 1). For the investigation of effects of heterogeneous and anisotropic waste, $H_w$ and $d_v$ are 17 and 3 m, respectively. The leachate injection boundary, which is the perforated casing of the VW, was simulated as a specified flux boundary with the flux values specified in the respective zones.

The model is discretized into cells and all of the external boundaries are simulated as zero flow boundaries. To simulate the leachate collection and removal system (LCRS) at the bottom, zero pressures are defined for all cells in the bottom layer of the model and the sum of outflow from these cells is calculated as outflow through the LCRS. Preliminary modeling by using different grid cell sizes found that a grid cell size of 0.3 × 0.3 m provides accurate results.

To investigate the effects of the relative screen locations (shallow versus deep; staggered versus uniform) and spacing between the VWs, a larger bioreactor landfill cell of 150 m length and 30 m height is considered. MSW was simulated in 10 layers, with each layer 3 m thick. The diameter and screen length of the VW are 0.3 and 3 m, respectively. The three different geometric configurations of VW (Fig. 2) with varying leachate injection screen locations and number of wells were investigated:

- **Configuration VW-C1** has five VWs, evenly spaced at 30 m. Each VW has a single leachate injection screen. Those screens are located at shallow or deep depths, identified as S-W and D-W, respectively, to represent the shallow and deep wells. The vertical distance of the injection screens, measured between the two adjacent VWs, is 12 m between the centers of the injection screen in each VW (Fig. 2). The total VW depths from the ground surface ($H_w$) for the shallow well and deep well are 7.5 and 19.5 m, respectively.

- **Configuration VW-C2** consists of three VWs with the horizontal spacing set at 60 m between the wells. There are two leachate injection screens in each VW, located at 7.5 and 19.5 m, measured from the ground surface to the centers of the screens (Fig. 2).
• Configuration VW-C3 consists of nine closely spaced VWs with a horizontal spacing of 15 m between each adjacent VW. This configuration uses one leachate injection screen in each VW. It is located at 7.5 and 19.5 m (measured to the center of the screen) for S-W and D-W, respectively, from the ground surface (Fig. 2), giving a vertical spacing of 12 m between screens of the adjacent VWs.

Thus, Configuration C1 has a total of five well screens, Configuration C2 has six screens, and Configuration C3 has nine screens.

Material Properties

Limited data are available on the saturated conductivity of MSW. Reddy et al. (2009) reported a laboratory study on the saturated hydraulic conductivity of MSW as a function of normal pressure; these data can be expressed by the following relationship:

\[ k_v = k_{vs} \left[ 1 + \left( \frac{\sigma_v}{p_a} \right) \right]^{-5.3} \]  

where \( k_{vs} \) = initial saturated hydraulic conductivity at zero normal stress; \( k_v \) = saturated hydraulic conductivity under effective normal stress of \( \sigma_v \); and \( p_a \) = atmospheric pressure.

Limited data have been reported on the unsaturated hydraulic conductivity of MSW based on laboratory testing. These studies model the relative hydraulic conductivity of MSW by using the van Genuchten model. The corresponding model parameters are reported in Table 1. The parameters reported by Haydar and Khire (2005) are not based on actual testing of MSW. Instead, they assumed the properties to be typical of silt loam. Breitmeyer and Benson (2011) and Stoltz et al. (2012) reported the parameters for various unit weights of MSW. The effect of variation in the hydraulic conductivity of MSW was investigated by using the bioreactor landfill cell model shown in Fig. 1 and by using five sets of reported parameters.

To investigate heterogeneous and anisotropic conditions, the MSW in the bioreactor landfill cell shown in Fig. 1 is modeled with: (1) homogeneous isotropic waste (HIW), (2) heterogeneous isotropic waste (HTIW), or (3) heterogeneous anisotropic waste (HTAW). These three conditions are modeled by varying the saturated hydraulic conductivity. The unsaturated hydraulic parameters are kept constant for all cases (HIW, HTIW, and HTAW) and follow the values given by Khire and Mukherjee (2007). As the leachate is recirculated, the outflow through the leachate collection and removal system, located at the bottom of the landfill, is computed. The degree of saturation (or the wetted area), pore water and gas pressures, and leachate outflow at LCRS were compared.

Boundary and Initial Conditions

All of the model boundaries were assumed to be impermeable. The grid points were initially free to vary based on the net inflow and outflow from the neighboring zones. Pore-water pressure was fixed to zero for the grid points at the bottom layers to represent the drainage layer. The base was also fixed and the lateral and vertical deformations of the cell at base are zero to compute the total outflow rate at LCRS. Furthermore, the lateral deformation was restrained on the sides of the model. The initial pore-water pressure values of wetting and nonwetting fluids were assumed to be zero at all grid points. For all modeling cases in this study, the residual saturation, initial saturation and initial porosity of MSW in the grid zones were assumed to be 25, 40, and 45%, respectively.

Model Simulations

Unsaturated hydraulic parameters for MSW, presented in Table 1, were used to simulate the effect of unsaturated hydraulic conductivity. Leachate is injected continuously at the rate of 25 m$^3$/day until steady state conditions are reached. This rate was selected as the average leachate injection rate consistent in previous studies (Khire and Mukherjee 2007). As the leachate is recirculated, the outflow through the leachate collection and removal system, located at the bottom of the landfill, is computed. The degree of saturation (or the wetted area), pore water and gas pressures, and outflow rate at LCRS were compared.

Once the system attains steady state, the effect of gravity drainage on the moisture distribution is investigated during four weeks after the cessation of leachate injection. The changes in the degree of saturation, pore water and gas pressures, and leachate outflow at

### Table 1. Published Unsaturated Hydraulic Properties of MSW

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Unit weight (kN/m$^3$)</td>
<td>7.0</td>
<td>—</td>
<td>—</td>
<td>5.5</td>
<td>4.5</td>
</tr>
<tr>
<td>Porosity</td>
<td>0.53</td>
<td>0.45</td>
<td>0.58</td>
<td>0.60</td>
<td>0.69</td>
</tr>
<tr>
<td>Matric suction, ( \alpha ) (1/kPa)</td>
<td>0.26</td>
<td>19.6</td>
<td>1.4</td>
<td>0.65</td>
<td>1.23</td>
</tr>
<tr>
<td>Residual moisture content, ( \theta_r )</td>
<td>0.11</td>
<td>0.067</td>
<td>0.0</td>
<td>0.21</td>
<td>0.22</td>
</tr>
<tr>
<td>Saturated moisture content, ( \theta_s )</td>
<td>0.53</td>
<td>0.45</td>
<td>0.58</td>
<td>0.60</td>
<td>0.53</td>
</tr>
<tr>
<td>van Genuchten ( n )</td>
<td>2.2</td>
<td>1.41</td>
<td>1.6</td>
<td>1.89</td>
<td>1.60</td>
</tr>
<tr>
<td>van Genuchten ( a )</td>
<td>0.55</td>
<td>0.29</td>
<td>0.37</td>
<td>0.47</td>
<td>0.375</td>
</tr>
<tr>
<td>van Genuchten ( m )</td>
<td>0.50</td>
<td>0.50</td>
<td>0.50</td>
<td>0.50</td>
<td>0.50</td>
</tr>
<tr>
<td>van Genuchten ( c )</td>
<td>0.50</td>
<td>0.50</td>
<td>0.50</td>
<td>0.50</td>
<td>0.50</td>
</tr>
</tbody>
</table>

Note: \( k_{sat} \) for HIW simulations = \( 1 \times 10^{-5} \) cm/s; \( k_{sat} \) for HTIW and HTAW conditions ranged from \( 1 \times 10^{-3} \) cm/s for the uppermost waste layer to \( 1 \times 10^{-5} \) cm/s for the lowermost waste layer based on Eq. (13); \( k_{sat}(h) \) for HTAW simulations was 10 times the \( k_{sat}(r) \).

\(^4\)Haydar and Khire (2005) assumed silt loam to represent MSW.
the bottom of the landfill during the gravity drainage are all computed. The results are compared for all sets of unsaturated hydraulic parameters, presented in Table 1.

To investigate the effect of heterogeneous and anisotropic waste conditions, three simulations are performed:

- For HIW, MSW is assumed to be a single homogeneous waste layer having isotropic hydraulic properties throughout the landfill mass.
- For HTIW, MSW is assumed to be filled in 10 layers that represent the inhomogeneity of the waste. The hydraulic conductivity of waste varies with depth. However, MSW has isotropic variation of hydraulic properties in each layer.
- For HTAW, the most realistic MSW model assumes varying hydraulic properties in each layer and anisotropic variation of $k_{sat}$ with horizontal hydraulic conductivity ($k_v$) 10 times the vertical hydraulic conductivity ($k_h$). This anisotropy ratio is consistent with that used by Haydar and Khire (2004).

These simulations are performed with continuous leachate injection to assess the transient leachate distribution until steady-state condition or 12 weeks are reached, whichever occurs first. The leachate is injected at three different flow rates, 5.5, 28, or 55 m$^3$/day, similar to the flow rates simulated by Khire and Mukherjee (2007), who assumed MSW as HIW material. The results of the HIW from this study are compared with the results reported by Khire and Mukherjee (2007).

The models shown in Fig. 2 are used to assess the effect of the three VW geometric configurations, denoted as VW-C1, VW-C2, and VW-C3. MSW was assumed to be heterogeneous and anisotropic (HTAW). In these cases, the leachate is continuously injected into the VWs at the rate of 55 m$^3$/day for four weeks.

The continuous injection of leachate is rarely practiced. Instead, intermittent leachate injections are repeated with cycles of a few hours of injection followed by gravity drainage. To investigate the effect of intermittent leachate injection, simulations are performed for heterogeneous anisotropic waste condition with a single VW for an injection rate of 55 m$^3$/day, injected at these two on and off frequencies: (1) one day on and one day off; and (2) one week on and one week off. The results are compared to the continuous leachate injection mode to discern the wetted area and pore water and gas pressures.

**Results and Discussion**

**Effect of Unsaturated Hydraulic Conductivity**

Fig. 3 compares the wetted area (m$^2$) for all sets of MSW unsaturated hydraulic properties (Table 1) after four weeks of leachate injection and at steady state. The wetted areas corresponding to a degree of saturation of 60% and more are presented in Fig. 3 for HIW simulations. The difference in the wetted areas for each set of hydraulic properties is relatively small. The wetted area after four weeks of leachate injection ranges approximately between 96 and 150 m$^2$ and the range for the wetted area at steady state is approximately 250 to 320 m$^2$ for the unsaturated hydraulic properties given in Table 1. The MSW wetted areas at steady state vary within 20% for the different sets of unsaturated hydraulic properties. Once the steady state is reached, the wetted area does not vary significantly, because the injected leachate is constantly collected from the drainage layer. This finding is consistent with the results of various studies, including that of Haydar and Khire (2005). On the other hand, because the relative permeabilities of porous media for wetting and nonwetting fluids are different for each set of hydraulic properties (Table 1), the rate of leachate migration in the landfill is affected and shows variations in the wetted MSW area after the first four weeks of leachate injection. This indicates that the unsaturated hydraulic properties influence leachate migration within the landfill for transient conditions.

Fig. 4 shows the outflow from the LCRS for various sets of unsaturated hydraulic properties for the HIW listed in Table 1. The outflow collected after four weeks of continuous leachate injection ranges from 4 to 22 m$^3$/day, and for steady state, it ranges from 24 to 24.8 m$^3$/day. The time to reach a steady state for the different sets of parameters varies greatly, from 42 to 150 days. The time required to reach a steady state varies for different sets of unsaturated hydraulic parameters of MSW, although the outflow rate is approximately the same at steady state, as expected and previously reported (Haydar and Khire 2005). The relative permeabilities vary significantly for different sets of unsaturated hydraulic parameters, which affects the time for the injected leachate to migrate and reach steady state. A dissimilar time period to reach steady state proves the importance of considering the effect of unsaturated hydraulic properties for transient conditions.

Fig. 5 shows the maximum pore water and gas pressures developed as a function of the duration of leachate injection and the flow rate of injection for two sets of unsaturated hydraulic properties. When the leachate recirculation is performed, assuming the unsaturated hydraulic properties of MSW given by Breitmeyer and Benson (2011) with MSW dry unit weight of 5.5 kN/m$^3$, a steady-state condition is attained in approximately 60 days; this is attained in 105 days with the unsaturated hydraulic properties given by Stolz et al. (2012) for MSW dry unit weight of 6.08 kN/m$^3$. Leachate recirculation ceases once the steady-state condition is reached and gravity drainage is allowed for four weeks succeeding the steady state. Evidently, the maximum pore-water pressure developed in the landfill gradually increases with the leachate injection; therefore, the gas pressure gradually drops to zero at steady state (with 100% saturation in MSW). During the gravity flow, the pore-water pressures drop, resulting in negative pore-water pressure after four weeks of gravity drainage. On the other hand, because of the decrease in the degree of saturation of the liquid phase, pore-gas pressures increase and reach magnitudes that are greater than the pore-water pressure (Fig. 5).

Fig. 6 illustrates the typical pore water and gas pressure profiles in the landfill, measured along the vertical well across the entire landfill.
height of the MSW. During the initial one week flow, the gas pressures are greater than the pore-water pressures. Further recirculation of leachate increases the degree of saturation; therefore, the water pressure increases, reducing the gas pressure. Fig. 6 also illustrates this effect for four weeks of leachate injection: as the degree of saturation of the MSW has risen, the pore-water pressure increases beyond the gas pressure. The leachate recirculation is continued until steady-state condition is reached. The levels of the pressure that develop are solely attributable to the leachate; therefore, it can be concluded that at 100% saturation, the pressure that develops is exclusively to the result of the presence of the leachate. The numerical modeling approach assumes the continuity of gas and liquid phases across the entire model domain. However, in the field, the continuity is seldom achieved. Hence, although the pressures presented in Fig. 6 are average magnitudes of gas and liquid pressures near midscreen location, much higher pressures may develop in disconnected voids. In addition, this modeling study does not consider the effect of gas generation and gas extraction within a landfill, which will influence the magnitudes of gas and liquid pressures. This study assumes that there is no net accumulation of landfill gas because of a 100% efficient gas extraction system.

Fig. 7(a) shows the pore-water pressure distribution plotted along the depth of the landfill near the injection screen of the VW at steady-state condition for \( Q_i = 25 \text{ m}^3/\text{day} \). The pore-water pressure that develops during the leachate injection is approximately the same for the three sets of unsaturated hydraulic properties and variation is minimal. This is because the MSW is saturated in all steady-state cases within the wetted area. The maximum pore pressure is 100–120 kPa for different sets of unsaturated hydraulic conductivity parameters at steady state. On the contrary, the pore-water pressure is different during the initial stages of the leachate injection period (Fig. 6).

To investigate the effect of gravity drainage on moisture distribution by using different sets of hydraulic properties, the gravity drainage (without leachate injection) is simulated for four weeks subsequent to the steady-state condition. Fig. 4 shows a decrease in outflow once the leachate injection ceases. The outflow that is collected after the four weeks of drainage of the leachate injection is stopped varies from 1 to \( 5 \text{ m}^3/\text{day} \). Thus, the unsaturated hydraulic properties influence the drainage durations and drainage flow rates.

Fig. 7(b) shows the vertical profiles of pore pressure distribution during the four weeks of gravity drainage. The results indicate that the pore pressure distribution is significantly different for each set of unsaturated hydraulic properties. It is evident that the unsaturated hydraulic properties affect the moisture distribution, especially if leachate injection is performed for some time, followed by gravity drainage. Higher negative pore pressure (\( \sim 11.6 \text{ kPa} \)) buildup is observed when the set of parameters suggested by Haydar and Khire (2005) is used. The variations observed in this study are based on different sets of unsaturated hydraulic properties, which are dependent on the composition, age, confining stress, and resulting pore size and structure of the MSW.
Effect of Heterogeneous and Anisotropic MSW

For HIW, the predicted wetted width of MSW (the area where the degree of saturation ≥ 60%) for injection rates equal to 5.5, 28, and 55 m$^3$/day is shown in Fig. 8. The results from Khire and Mukherjee (2007) for identical model input parameters are also shown; there is good agreement between the present study and the previously reported study.

The steady-state flow condition was achieved at different points in time, depending on the leachate injection flow rates. The time required to reach the steady-state flow condition is 44, 48, and 61 days for $Q_i = 55$, 28, and 5.5 m$^3$/day, respectively. This finding is consistent with the results of Khire and Mukherjee (2007).

MSW is a heterogeneous and anisotropic material. Therefore, to systematically investigate this condition, modeling is performed that uses two different conditions: HTIW and HTAW, and the results are compared with HIW conditions. When leachate is injected, the steady-state flow condition is reached within 11 weeks in the case of HIW. However, in the cases of HTIW and HTAW, the steady state is not attained, even after continuous leachate recirculation for 12 weeks, with one exception, in which the steady state is observed only for the low leachate injection rate of 5 m$^3$/day. Because of inhomogeneity, the wetted width of MSW gradually increases. For the three simulated conditions, the model results for saturation levels, wetted width, pore-water pressure, and outflow flux are compared. Fig. 9 illustrates the differences in saturation contours (wetted area) with different waste conditions for a leachate injection rate ($Q_i$) of 55 m$^3$/day. The results show that the lesser permeability of the lower MSW layers results in greater lateral migration of the leachate, which leads to the higher wetted width reported for HTIW, as shown in Fig. 8. Haydar and Khire (2004) simulated heterogeneity and anisotropy and the results were consistent with this study. Khire and Mukherjee (2007) and Reinhart et al. (2002) also reported larger values of wetted width for lower permeability of MSW when MSW is considered as single layer. Because of the anisotropy in HTAW ($k_h = 10k_v$), the leachate tends to flow more in the lateral direction than in the vertical direction.

**Fig. 6.** Pore water and gas pressures as a function of time and depth for unsaturated hydraulic properties of MSW, represented by Stoltz et al. (2012) for MSW $\gamma_d = 6.08$ kN/m$^3$ and injection rate of 25 m$^3$/day: (a) after one week of injection; (b) after four weeks of injection; (c) at steady state

**Fig. 7.** Distribution of pore-water pressure: (a) at steady state; (b) after four weeks of gravity drainage for three sets of unsaturated hydraulic properties of MSW (homogeneous isotropic case).
For all three waste conditions, the maximum saturation is 100% because of the relatively low permeable characteristic of MSW (in the order of $10^{-4}$ cm/s). The computed wetted widths are 21.8, 27.6, and 39.5 m for HIW, HTIW, and HTAW conditions, respectively, under the high injection rate of 55 m$^3$/day (Fig. 8).

The maximum pore-water pressure and gas pressure developed in the MSW for the homogeneous isotropic, heterogeneous isotropic, and heterogeneous anisotropic waste conditions, are shown in Figs. 10–12 at different depths near the VW. This is done for the leachate injection rate of 28 m$^3$/day. The results are shown for one week’s flow and four weeks’ flow, and for the steady-state condition or 12 weeks, whichever occurs first. These results illustrate the evolution of the pore water and gas pressures. The values for the pore pressures that developed at the different locations above and below the leachate injection are compared. This shows that, over time, as the leachate injection is continued, the gas pressure decreases. The gas pressure is dominant during the initial first week flow and is higher than the water pressure in the MSW. Further recirculation of the leachate results in an increase in the degree of saturation of the MSW, so as the water pressure increases, the gas pressure decreases. Thus, only pore-water pressure exists in the MSW when the steady-state condition is reached. All simulation results indicate that the pore-gas pressure never exceeds the pore-water pressure observed at steady state.

Fig. 13 shows the pore-water pressure distribution at different depths near the VW location for different injection rates in different MSW conditions at steady state or 12-week flow. For the HIW, the maximum pore-water pressure at steady state is approximately 170 kPa near the injection screen. Because of the low permeability in the bottom layers, in the case of the HTIW, the pore-water pressure is as high as 170 kPa after 12 weeks of flow near the vicinity of injection screen for the high injection rate. On the other hand, for HTAW, which is the more realistic case because of the anisotropic variation in hydraulic properties, the pore-water pressure is dissipated in the lateral direction (maximum pore-water pressure is approximately 50 kPa for $Q_i = 55$ m$^3$/day).

**Effect of Geometric Configuration**

Configurations VW-C1, VW-C2, and VW-C3, shown in Fig. 2, are compared based on the evolution of moisture saturation during the continuous leachate recirculation performed at the rate of 55 m$^3$/day for four weeks with HTAW. During the first two weeks of injection in VW-C1, the wetted area is relatively small. Hence, the leachate injection is simulated for four weeks. Even after four weeks, dry zones remain between the adjacent VWs in the landfill. Because the shallow MSW layers possess higher permeability than the deep layers, the shallow waste has lower moisture retention. Hence, the injected leachate tends to move down to the deep layers, leaving dry conditions in the shallow layers. This results in a non-uniform distribution of injected leachate in VW-C1. In the case of VW-C2, in which each VW is comprised of both shallow and deep injection screens, during the initial two weeks of leachate injection the saturation contours indicate that there is dry MSW in between
the VWs in the landfill cell throughout the depth of the landfill. The leachate injection is continued for the third and fourth weeks, and the results demonstrate that the uniform moisture distribution cannot be achieved in the VW-C2 configuration. The presence of the dual screens in each VW generated saturation overlap only in the vicinity of the VWs. As a result, a major portion of the MSW situated between the VWs remains dry. When there is heterogeneous MSW, the injected leachate migrates quickly toward the deep layers, but the less permeable MSW found in the deep layers slows down the further migration of the leachate. The injected leachate tends to move laterally in those deep layers because of the anisotropic properties of MSW. Configuration VW-C3, in which the horizontal spacing between the VWs is 15 m, increases the moisture distribution in the landfill compared to the configurations of VW-C1 and VW-C2 within the first two weeks. The comparisons of the third week flow for the three VW configurations indicate that VW-C3 is capable of achieving uniform distribution in the bioreactor landfill in as short a time as three weeks, compared to the other configurations. Furthermore, these results indicate that the injected leachate is uniformly distributed throughout the landfill area when the leachate injection is continued for four weeks. This leads to the conclusion that reducing the horizontal spacing between VWs increases the efficiency of the leachate recirculation system (Khire and Mukherjee 2007).

Fig. 10. Pore water and gas pressures for HIW as a function of time and depth for injection rate of 28 m$^3$/day: (a) after one week of injection; (b) after four weeks of injection; (c) at steady state

Fig. 11. Pore water and gas pressures for HTIW as a function of time and depth for injection rate of 28 m$^3$/day: (a) after one week of injection; (b) after four weeks of injection; (c) at steady state

Fig. 14 compares the wetted areas, expressed as a percent of total landfill area for the three VW configurations. In VW-C1, the percent of wetted area is as low as 9% after the first week of leachate injection. The wetted area increased to 18, 22, and 44% after two, three, and four weeks of leachate injection, respectively. This implies that more than half of the MSW in landfills does not gain moisture, even after four weeks of injection. In the case of VW-C2, the percent wetted MSW area is 10, 20, 25, and 35% after one, two, three, and four weeks of leachate recirculation, respectively. Although VW-C2 has multiple leachate injection screens in each VW, it produces nonuniform moisture distribution in the landfill. Moreover, the wetted area in VW-C2 is less than VW-C1 after four weeks of injection. Because the injected leachate migrates vertically down to the LCRS instead of migrating laterally to other areas, even less of the MSW area is saturated than in VW-C1. This implies that neither VW-C1 nor VW-C2 can produce uniform moisture distribution in the landfill. In the VW-C3 configuration, the results are more successful than with either VW-C1 or VW-C2. Here, the horizontal spacing between each VW is reduced and the results for the percent of wetted MSW area increase to 18, 30, 72, and 84%, respectively, following the first, second, third, and fourth week of leachate injections. Evidently, more than 70% of the landfill area is covered with the injected leachate within three weeks, and this percentage grows with time and continued...
injection. This indicates that VW-C3 is capable of effectively distributing injected leachate in the bioreactor landfill cell. Khire and Mukherjee (2012) showed that maximum wetting efficiency is achieved when the well spacing is 10 to 15 m. Hence, the findings of this study are consistent.

Understanding the pore-water pressure distribution in the landfill will help to regulate the leachate recirculation and ensure safe bioreactor reactor landfill operations. Fig. 15 shows the maximum pore water and gas pressures in the landfill for all three VW configurations over four weeks of leachate injection, which are observed near the leachate injection screens. The maximum pore water and gas pressures develop around the injection screen throughout the recirculation period, and the pore-water pressure is observed to increase while the gas pressures decrease. The gas pressures in the MSW are considered to be relatively high in all three VW configurations during the initial period of leachate recirculation, with maximum pore water and gas pressures observed after the first week of leachate injection at approximately 38 and 60 kPa, respectively, for VW-C1. In the subsequent weeks of leachate recirculation, the degree of saturation increases; therefore, the pore-water pressure values increase. These are compared to the gas pressure values (after two weeks of leachate recirculation). Further leachate recirculation during the fourth week results in an increase in the pore-water pressure to 120 kPa.

Because of the higher permeability of the shallower layers, the pore-water pressure is relatively small (~11–22 kPa), compared to that found in the deeper layers (pore-water pressure ~12–44 kPa), which possesses lower permeability. Most parts of the MSW in the landfill experience a pore-water pressure ranging between 11 and 33 kPa.

VW-C2, where each VW has dual leachate injection screens, shows a higher increase in the pore-water pressure than found in VW-C1. Similar to VW-C1, the maximum pore-water pressure is less than pore-gas pressure during the initial first week of leachate recirculation [Fig. 15(b)]. The maximum pore water and gas pressures in the landfill modeled as VW-C2 are approximately 42 and 53 kPa, respectively, near the injection screen. Because of the increase in the degree of saturation, the pore-water pressure that developed is higher than the pore-gas pressure after two and three weeks’ flow. The leachate injection, when continued for three weeks, increases the pore-water pressure to 70 kPa and reduces the pore gas pressure to 23 kPa. At the four-week mark, the pore-gas pressure is zero and the pore-water pressure is maximum. The maximum pore-water pressure after four weeks’ flow is approximately 90 kPa (compared to 120 kPa found in
VW-C1. This increase in pore-water pressure is explained by the saturation overlap in the vicinity of each VW.

Similar to VW-C1 and VW-C2, the gas pressure that develops in VW-C3 is higher during the first week of leachate recirculation and declines during the subsequent weeks of leachate injection as the waste approaches saturation [Fig. 15(c)]. In VW-C3, the pore water and gas pressures are approximately 35 and 56 kPa, respectively, during the first week of injection. Additional leachate recirculation reduces the pore gas pressure to 30 kPa (with a pore-water pressure of 42 kPa) after two weeks of flow. Continuous leachate recirculation through four weeks of flow increases the pore-water pressure, but gas pressure values are not observed. The observed pore-water pressure is 55 kPa after three weeks and 58 kPa following four weeks of leachate injection [Fig. 15(c)]. Higher pressure buildup occurs only near the leachate injection screen in the DW, which may be attributed to the known low permeability of the MSW in the deeper MSW layers. Most of the area of the landfill experiences pore-water pressure in the range of 16–48 kPa. In general, lower gas and water pressures are observed for simulations in which waste is heterogeneous and/or anisotropic due to lateral spreading of leachate. These results are consistent with the results of haydar and Khire (2004).

**Effect of Intermittent Injection**

Although steady-state simulations were conducted to provide the maximum wetted areas and to calculate maximum liquid pressures for design, achieving steady state in the field is unrealistic. Often, leachate injection is conducted in on and off dosing cycles (Haydar and Khire 2005; Mukherjee and Khire 2012), depending upon site-specific conditions such as leachate generation rate, temporary leachate storage, and leachate treatment and disposal options. Hence, intermittent leachate injection is simulated for HTAW conditions with a single VW for an injection rate of 55 m³/day, injected at these two on and off frequencies: (1) one day on and one day off; and (2) one week on and one week off. The resulting wetted areas for degree of saturation ≥60% are presented in Fig. 16. The cumulative volume of leachate injected during the 30 days of modeling is also presented in Fig. 16.

Fig. 16 shows that the wetted width gradually increases for all frequencies of leachate injection and none reach the maximum possible area at steady state within 30 days. The wetted area for the one day on/off simulation is slightly greater than that for one week on/off (with seven days on, seven days off, and the remaining nine days off, for a total of 30 days). Although the cumulative leachate volumes injected into the VW for these two simulations are approximately the same, if a longer duration is allowed for drainage after the injection is turned off, the overall wetted area grows at a slower rate. Fig. 4 shows that the typical drainage times for the saturated hydraulic conductivities selected in the model are approximately two to six days. Hence, if leachate injection is turned off for greater than two days, the initial degree of saturation of the waste drops significantly and the wetted areas grow at a relatively slow rate with intermittent injection.

Fig. 17 shows the effects of continuous and intermittent leachate injection on the evolution of pore water and gas pressures for continuous injection at 55 m³/day. The pore pressures at a point immediately next to the midscreen of the well are shown in Fig. 17(a). The maximum liquid pressure at the end of the 30 days is approximately 160 kPa, whereas the gas pressure, which peaked at 60 kPa, drops to 10 kPa after 30 days. As the degree of liquid saturation of the waste increases, the gas pressure decreases. For intermittent injection of leachate, the liquid pressures gradually increase while the gas pressures also increase near the mid-screen location of the well, because it is surrounded by relatively permeable material, which drains relatively quickly. Consistent with
the experimental findings of Mukherjee and Khire (2012), liquid pressures drop immediately after the injection stops.

Fig. 17(b) shows the pore water and gas pressures at a depth equal to a meter below the bottom of the well. All of the liquid and gas pressures are lower than those generated at the middepth of the well screen. With continuous injection for 30 days, the maximum liquid and gas pressures are generated at 95 and 10 kPa, respectively [Fig. 17(b)]. The variation of liquid and gas pressures for intermittent leachate injection mode is similar to that found at the middepth of the well screen. When the injection rate is 1 day on/off, the gas pressures reach approximately 70 kPa after 30 days, whereas the gas pressures for one week on/off reach approximately 40 kPa below the well. The liquid pressures are substantially reduced with one week on and off cycles, with 40 kPa of gas pressure and 10 kPa of liquid pressure. Thus, intermittent liquid injection, in which the waste is highly permeable and cannot be wetted for extended periods of time, will result in higher pore-gas pressures.

**Conclusions**

Moisture distribution in a bioreactor landfill subjected to leachate injection from VW was simulated by using a numerical two-phase flow model, FLAC. The numerical model was validated by using published numerical studies. The effect of unsaturated hydraulic properties of MSW was evaluated first, followed by the effect of heterogeneity and anisotropy of MSW. Finally, the study simulated the effect of three geometric formations of VWs (spacing and configuration) on moisture distribution in a bioreactor landfill cell. Continuous and intermittent liquid injection scenarios were simulated and the pore water and gas pressures were assessed. The following conclusions can be drawn from this study:

- Unsaturated hydraulic properties significantly influenced the moisture distribution (wetted area, wetted width, or saturation levels), pore water and gas pressures, and the leachate outflow rate during and immediately after leachate recirculation for
transient conditions. The time to reach steady-state flux is governed by the relative permeabilities of the wetting and non-wetting fluids.

- Simulations for heterogeneous and anisotropic MSW conditions showed significant differences in the saturation levels, wetted width, pore water and gas pressures, and outflow rate compared to homogeneous and isotropic conditions. Comparisons demonstrate that steady-state flow condition was achieved in the HIW condition in less time than in HTIW and HTAW waste conditions. Furthermore, the wetted width considerably increased in the HTAW compared to HIW conditions because of the gradual decrease in the vertical saturated hydraulic conductivity and/or greater horizontal conductivity.

- Staggered geometry with closer spacing of VWs achieved relatively uniform moisture distribution in the landfill. As the spacing between the wells decreased, the degree of saturation, pore pressure developed for both water and gas, and the outflow collected at leachate collection and removal system all increased.

- The magnitude of pore-gas pressures during intermittent leachate injection would be greater than the pore-water pressures in regions of relatively high permeability. The gas pressures would continuously increase or decrease, depending upon the duration of on/off dosing cycles and location in the waste with respect to the well. Hence, not only the pore-water pressures, but gas pressures, should be considered for the evaluation of the slope stability of bioreactor landfills, especially when the efficiency of gas extraction systems is less than optimal.

Acknowledgments

This project is funded by the U.S. National Science Foundation (CMMI #0600441) and it is gratefully acknowledged.

References


