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Causes of high internal pore-pressures in a downward-draining MSW landfill

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ABSTRACT

A two-phase liquid/gas flow numerical model has been used to investigate the presence of elevated porewater pressures in a 20-metre deep landfill underlain by a fully drained leachate collection layer. Monitoring of leachate levels in the landfill using piezometers located at different discrete levels within the landfill found water table type conditions to within 10 metres of the surface and a strong downward hydraulic gradient at an infiltration rate of 400 mm/year.

Short duration falling head tests in piezometers indicated landfill hydraulic conductivities (K) between $1x10^{-4}$ and $1x10^{-5}$ m/s, with a general reduction in K with depth.

Several different hypotheses that could explain the high porewater pressures in the landfill were investigated using a one-dimensional configuration of the landfill degradation and transport numerical model LDAT. It was assumed that the unsaturated properties of the landfilled wastes can be bounded by two sets of van Genuchten parameters.

Comparing the values of K_v required to create a match between observed and modelled leachate heads with the measured K_h values at the site, leads to a tentative conclusion that landfill scale anisotropy could be as high as ~1:1000 at the study site (i.e. K_h approximately 3 orders of magnitude higher than K_v).

The introduction of a distributed landfill gas (LFG) source term into LDAT at a rate of 0.61 $m^3_{LFG}\ t_w^{-1}\ yr^{-1}$, similar to the gassing rate at the site, increased the adopted permeability relationship in LDAT by a factor of between ~3 and ~7.5 compared with a no gassing scenario.

Introducing even moderate gas generation rates (5.6 $m^3_{LFG}\ t_w^{-1}\ yr^{-1}$) into models simulating a reduced infiltration rate of 50 mm/ year can result in a significant depth of waste where porewater pressures are more than 1 kPa (10 cm water head). This results in apparent below water table type conditions as water will enter piezometers installed into such wastes, even though the gassing reduces the degree of saturation to below 1.

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Introduction

Background

Most modern landfills are built with a leachate drainage system on top of a low permeability liner to facilitate the control of leachate within the site. Typically, these basal drainage systems comprise a layer of high permeability drainage material (gravel, coarse sand or tyres) over the whole of the site base, with a network of drainage pipes for the collection and removal of leachate. The primary goals of such systems are to: 1) enable control of the leachate head transmitted onto the basal liner, 2) reduce pore water pressures within the landfilled waste, for example to improve waste stability, and 3) control or reduce the degree of saturation of the waste, for example to aid landfill gas extraction (which is not generally possible from saturated waste). Understanding the performance of basal drainage systems is also critical to the design and operation of leachate recirculation, flushing, and aeration landfill management technologies.

There has been considerable research on the hydraulic performance of leachate drainage systems with respect to clogging (e.g., Paksy et al., 1998; Fleming and Rowe, 2004; VanGulck and Rowe, 2008; Beaven et al., 2013), but little on the hydraulic interaction between the drainage layer and the overlying waste. Powrie and Beaven (1999) calculated a theoretical leachate head profile within a landfill for one-dimensional downward flow where the hydraulic conductivity of the waste reduced with increasing effective stress. The recharge rates needed to create saturated conditions in the landfill were determined for various vertical hydraulic conductivity profiles and a fully drained basal drainage system (i.e. no head on the liner). Recharge rates of ~1 m/day were required to generate leachate heads in landfills where there had been no pre-compaction of the waste; as a result of the dependence of hydraulic conductivity on effective stress, there was a rapid increase in leachate head with elevation directly above the waste drainage layer interface.

The research underpinning the current paper was prompted by the discovery of elevated leachate heads of several metres above the base of the Kragge landfill, the Netherlands, which is operated with a completely drained leachate collection system. The Kragge landfill is part of the Dutch sustainable landfill management project (RIVM, 2014) where leachate recirculation is being undertaken. Recharge rates are estimated to be between 100 and 700mm/year (3.2×10^{-9} to 2.2×10^{-8} m/s). Preliminary in situ falling head tests in wells with discrete monitoring zones within the landfill indicated that the bulk hydraulic conductivity of the waste was in the range, or greater than, that used by Powrie and Beaven (1999); hence the extent of saturation in the waste was unexpected. There is a need to understand the mechanisms behind the large leachate heads in the landfill before proposing technical and practicable solutions that would reduce them. This study has relevance to other landfills as high leachate heads have been reported to occur in many landfills with basal collection systems, with high organic waste landfills being especially problematic in this regard (e.g., Zhang et al., 2013; Zhan et al., 2015). The study has also been extended to consider the implications on pore water pressures and leachate heads in containment landfills with much lower infiltration rates.

At a simple vertical one-dimensional level, for a build-up of leachate head to occur in waste overlying a fully drained basal leachate collection system, the average vertical hydraulic conductivity of the waste must be less than the vertical infiltration rate. Theoretically, a thin low permeability layer at or just above the interface between waste and drainage layer (for example, caused by clogging, especially if a geotextile has been used) could result in the same *maximum* leachate head in the site as that caused by the whole waste mass having a hydraulic conductivity (K) just less than the infiltration rate. This is because the overall vertical hydraulic conductivity of a soil or a series of layers whose hydraulic conductivity varies with depth is given by the overall harmonic mean conductivity or the thickness-weighted harmonic mean of the individual layers (e.g. Bear, 1972), and hence is dominated by the lowest conductivity over the whole vertical profile. However, the distributions of head with depth within the waste in these two cases would be quite different.

There are two key potential mechanisms that could result in a reduction in waste hydraulic conductivity, either generally throughout the waste mass or more locally in the vicinity of the drainage layer.

1. The layered nature of waste is known to lead to anisotropy in the hydraulic conductivity of waste, with reported horizontal to vertical ratios (K_h:K_v) covering a wide range of values. Hudson et al., (2009), in tests on large (2 m diameter) samples of waste in a compression cell, found K_h:K_v ratios between 5:1 and 10:1, with anisotropy increasing with effective stress. Similar results have been reported by other authors, including Ke et al., (2017). Laboratory tests on MBT (Mechanically Biologically Treated) waste by Munnich et al., (2005) indicated an upper bound hydraulic conductivity ratio of 250:1. In the field, the anisotropy that arises from layering of the waste is likely to be exacerbated by macro-scale effects arising from the placement in landfills of soils and daily covers. This may be reflected in the field scale tests of Singh et al., (2014), who reported anisotropy ratios of landfilled wastes of up to 100:1.
2. *In situ* landfill gas generation can cause otherwise fully saturated waste to become unsaturated (e.g., Merry et al., 2006), which will result in a reduction in hydraulic conductivity. Even if gas generation rates are low, gas accumulation within landfilled waste may affect its liquid phase hydraulic conductivity, especially at low pore-water pressures (Powrie et al., 2008).

Anisotropy, heterogeneity (i.e. the presence of identifiable layers of lower hydraulic conductivity) and desaturation of the waste could all impede the vertical flow of leachate into a basal drainage system. The aim of this paper is to investigate these factors in terms of their potential to cause elevated pore-water pressures in landfilled wastes directly overlying a fully functioning leachate collection system. The objectives of the paper are framed in terms of a series of different hypotheses, each of which are tested where possible against field data from the landfill.

Hypotheses investigated

Based on the information summarised above, we propose the most likely causes of high leachate heads in landfills underlain by a fully functioning leachate collection system may be:

1. A lower vertical hydraulic conductivity than that expected from the literature and field data, potentially caused (but not explicitly modelled) by one or more of the following:
    a. Vertical to horizontal hydraulic conductivity anisotropy in the waste, which could be widened to include landfill scale anisotropy caused by daily cover layers
    b. Trapping and accumulation of historically-produced gas in pockets (e.g. Hudson et al., 2009)
2. A thin, low hydraulic conductivity interface layer caused by:

   a. Physical or biochemical clogging at or near the interface of the waste and drainage layer
   b. Possible partial dewatering of the waste directly overlying a fully drained drainage layer, leading to a localised reduction in hydraulic conductivity in the waste associated with a combination of higher effective stress and/or desaturation.
3. Ongoing landfill gas production causing a reduced degree of water saturation and hence a lower hydraulic conductivity of the waste.

The above hypotheses are tested in the context of the average vertical pore water pressure profile measured at the case study site.

Relevance of unsaturated flow theory

Hypotheses 1b, 2b and 3 above depend on unsaturated flow hydraulics. The water retention curve (WRC - the relationship between capillary pressure and water content) and the relative permeability K(θ) function (how K changes with water content (θ)) are required for modelling multiphase and unsaturated flow in porous media to obtain the pore-pressure distributions in the liquid and gas phases. Original work on unsaturated flow in natural materials (e.g., Haines, 1930; Brooks & Corey, 1964; Gardner, 1937) has resulted in a large repository of experimental data on unsaturated soils (e.g., Nemes et al., 2001; Wosten et al., 1999). For use in unsaturated flow modelling, these data are fitted to empirical constitutive functions to describe the WRC and the K(θ) relationship in mathematical terms. The most commonly used functions are those of Brooks and Corey (1964) and van Genuchten (1980). Both allow matching with experimental data through a number of WRC fitting parameters. Importantly, the same WRC fitting parameters are also used in the relative permeability (K(θ)) constitutive function, which are difficult and time-consuming to measure experimentally (e.g. Benson and Gribb, 1997).

The widely-used Mualem / van Genuchten (1980) constitutive equations are adopted in this paper:

$$\Theta = \frac{(\theta - \theta_r)}{(\theta_s - \theta_r)} = [1 + (\alpha\psi)^n]^{-m} \quad (1)$$

and

$$K_\theta = K_s \Theta^l \left[1 - \left(1 - (\Theta)^{1/m}\right)^m\right]^2 \quad \text{(after Mualem, 1976)} \qquad (2)$$

where

$\Theta =$ effective saturation (often denoted S_e as in this paper)

$K_\theta =$ Hydraulic conductivity at water content θ

$K_s =$ Saturated hydraulic conductivity

$\psi =$ matric suction (capillary pressure $= p^G - p^L$, where p^G and p^L $=$ pressure of gas and liquid)

$\theta =$ volumetric water content at matric suction ψ

$\theta_s =$ saturated volumetric water content

θ_r = residual volumetric water content

α = fitting parameter (reciprocal units of pressure to that used in ψ)

n = fitting parameter

m = fitting parameter $= (1-(1/n))$

l = fitting parameter (often $= 0.5$)

The van Genuchten formulation involves just two fitting parameters when the WRC curves are plotted in terms of S_e. Consideration of naturally occurring soils ranging from clayey silts to coarse grained sands shows that values of α typically vary from ~0.1 to 3.5 kPa^{-1}, and values of n from 1.05 to 2.5. There is also a correlation between values of n and hydraulic conductivity, with low permeability silts and clays having low values of n.

The Mualem/van Genuchten unsaturated K_θ function, equation (2), requires the fitting value m (which is commonly linked to n) and l. The parameter n has the largest impact on how rapidly unsaturated hydraulic conductivity reduces with reducing degrees of saturation (see also supplementary information). Thus it might be inferred that waste characterised by a low value of n is more likely to support hypotheses 2b and 3 above.

Experimental data on WRC for MSW have been reported in studies starting with Korfiatis et al., (1984), and collated by Beaven et al., (2011), White et al., (2015) and Breitmeyer et al., (2020). Early published data on modelling of the unsaturated flow properties of wastes was generally predicated on a lower bound value for the van Genuchten fitting parameter n of ~1.4 (e.g. Staub, 2010; Haydar and Khire, 2005); however, this was probably affected by the difficulties of unsaturated flow testing at high effective stresses.

Powrie (McDougall et al., 1998) first surmised that the van Genuchten parameters for waste are likely to change with depth in a landfill; subsequent investigations including by Stoltz and Gourc (2007), Hossain et al., (2009), Stoltz et al., (2012), White et al., (2015) and Breitmeyer et al., (2020) have supported this. Breitmeyer et al., (2020) established clear relationships between α and the dry unit weight of MSW in laboratory specimens, with α decreasing with increasing dry unit weight. Their work also suggested that the parameter n varied between 1.2 and 1.6 over the range of dry unit weights measured in field scale experiments.

Method

The hypotheses are investigated using the numerical model LDAT (Landfill Degradation and Transport model, White et al., 2003; LDAT, 2022). The modelling is intended to have general applicability, but the results are calibrated with reference to monitoring data from the particular case study that prompted the research.

Field methods

Landfill site description

Cell 3 at the Kragge landfill is a fully lined above-ground land-raise containing ~1 million tonnes of mixed household and non-hazardous commercial wastes, landfilled between about 1999 and 2008. The cell is 5.6 hectares (~150 m × 350 m on plan) with a maximum depth of ~20 metres. The site is underlain by a 2 mm thick HDPE liner below a 300 mm thick basal leachate drainage layer comprising gravel. Six parallel 125 mm HDPE perforated pipes run through the drainage layer at a spacing of 25 m, allowing gravity drainage of leachate to a single Cell 3 collection and pumping sump. Leachate recirculation is undertaken on a daily basis into 14 parallel injection trenches at 20 m spacing on the top of the landfill, adding to the effective rainfall that infiltrates the waste.

The vertical infiltration entering the Cell 3 basal drains is a matter of some debate. The total leachate volume removed from the Cell 3 sump equates to an aerially distributed annual recharge rate of ~680 mm /year, but this volume includes leachate discharging from seepage faces on the flanks of the landfill and some surface water runoff (i.e. it is not all from vertical flow into the basal drains). A starting assumption is that vertical infiltration into the basal drains is ~ 400 mm/ year (although a much lower value of ~100 mm/year has not been ruled out).

The average landfill gas (LFG) generation rate in Cell 3 is estimated from site operator records as being ~ 0.033 m^3_{LFG} m^{-2} d^{-1} or 0.04 kg_{LFG} m^{-2} d^{-1}. Based on an average landfill depth of 20 m, and an average wet waste density of 1 tonne / m^3, this is equivalent to a gas generation rate of 0.61 m^3_{LFG} t_w^{-1} yr^{-1}. For comparison, the Environment Protection Agency's (EPA) Landfill and Landfill Gas Energy database for August 2022 was analysed (EPA, 2023). Data for total waste tonnages and current landfill gas extraction rates are presented for over 1300 landfills/ landfill cells. The average landfill gas extraction rate was 3.9 m^3_{LFG} t_w^{-1} yr^{-1} (STD 2.6) with maximum and minimum values of 47 and 0.2 m^3_{LFG} t_w^{-1} yr^{-1}. The LFG generation rate for Cell 3 reported here is at the lower end of this range, reflecting the extent to which biological stabilisation at the site has already occurred.

Piezometric measurements

Piezometric levels were obtained from a network of 23 piezometers, each with a 2 m response zone located at elevations between 6 m and 14 m above the base of the site. Continuous pumping from the basal drainage layer ensured that the head within it stayed low, varying between 0 and 1 m above the base of the site. Figure 1 shows an interpretation of the pore water pressures measured at elevations between 7 and 14 m above the base of the site. The plotted values are derived from average water level readings at a number of piezometers at similar depths. The readings have remained relatively unchanged over a period of at least 3 years and appear to represent effectively steady state conditions in the site. Two possible interpolations of water pressures between the values measured in the basal drain and at an elevation of about 7 m are indicated, together with a theoretical hydrostatic pressure distribution corresponding to no vertical drainage and fully saturated conditions to a depth of 14.2m.

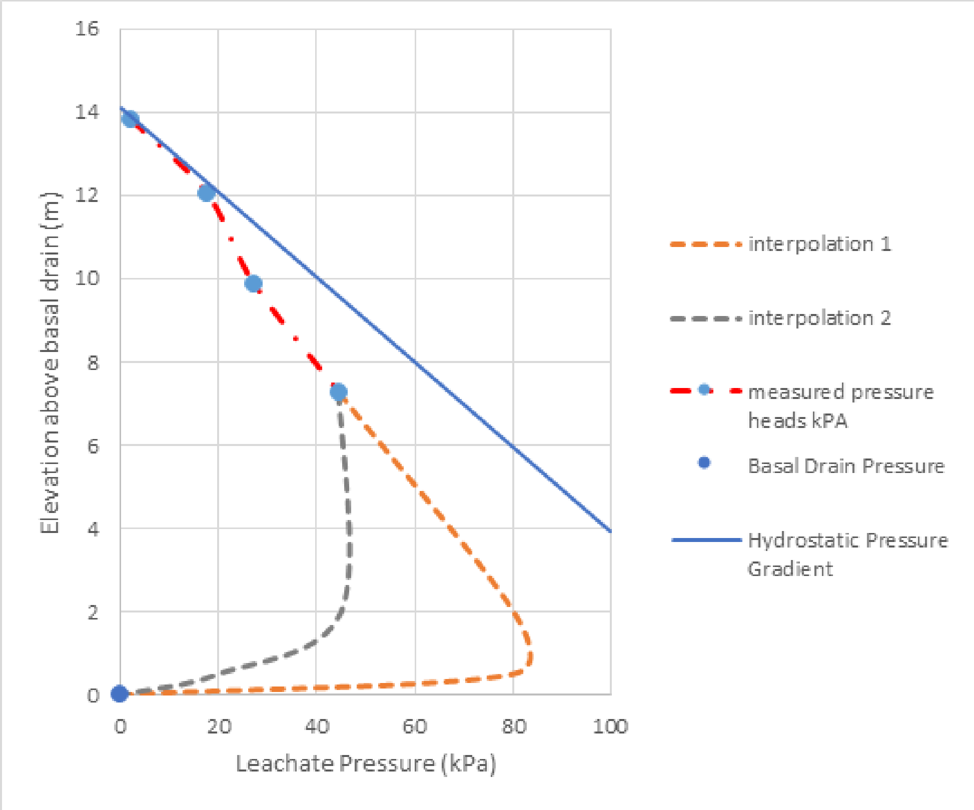

Figure 1 Average pore-water pressure measurements at discrete monitoring elevations at study landfill

The measured data are close to hydrostatic near the top of the waste. The pore pressure profile gradients then fall below hydrostatic before becoming zero and then negative, as the pressure returns to zero in the basal drain.

This is consistent with expectations in a saturated waste where hydraulic conductivity *k* reduces with depth, as proposed and demonstrated by Powrie & Beaven (1999). A pore pressure profile of this shape gives an increasing hydraulic gradient *i* with depth, such that the downward flowrate per unit area $q/A = k.i$ remains constant.

Later in this paper, results from different numerical simulations are compared with the two vertical pressure head distributions interpolated from the site data.

Falling head hydraulic conductivity tests

Six falling-head tests were carried out in monitoring piezometers (installed ~3 years prior to the tests) to estimate horizontal hydraulic conductivity. The piezometer boreholes were drilled using a sonic-drilling displacement technique and installed with 30 mm plastic pipe. The lower 1 m or 2 m of the pipe was perforated. No filter pack was provided, and the wells were not backfilled as the installation design relies on the waste closing back around the plastic pipe. The horizontal spacing between the piezometers was ~25 m.

Tests were carried out by injecting water into the top of the well at a constant rate of ~21 litres / minute for between 60 and 70 seconds. The head in the piezometer was measured using an automated pressure transducer prior to, during and after injection until the head had returned to its initial level.

Recovery to pre-test levels took between 15 and 90 minutes. The results were analysed using a simple model based on the Bouwer-Rice (1976) method, least-squares fitted separately to injection and recovery data.

The narrow diameter of the piezometers, the absence of a filter and the method of water injection are not ideally suited to a linear analysis. Nonetheless, the results give an order-of-magnitude estimate of horizontal hydraulic conductivity, which was repeatable and consistent between piezometers. The trend of variation in horizontal hydraulic conductivity (K_h) with depth is compared with estimated vertical hydraulic conductivities (K_v) based on the relationships between K and effective stress in Powrie & Beaven (1999) and Beaven (2000) for a fresh MSW type waste (PB MSW) and an aged degraded MSW type waste excavated from a landfill (PB Aged Landfill - Figure 2.). In this series of tests Powrie and Beaven (1999) used a large scale (2-m diameter) uniaxial compression cell to establish the hydraulic properties, (including vertical hydraulic conductivity) of different waste types under applied stresses up to 600 kPa. As already summarised, K_h is anticipated to be larger than K_v which, to a large extent, Figure 2 indicates. The differences between K_h and K_v could be explained by waste anisotropy (e.g., Hudson et al., 2009; Ke et al., 2017) as previously discussed.

Also plotted on Figure 2 are the range of potential infiltration rates reaching the basal drain (between 100 and 700 mm/year) converted to an equivalent vertical flow in m/s. The implication of this range of values is that, in theory, to avoid a build-up of saturated condition within the landfill, the infiltration rate has to be less than the average K_v. From the measured and literature values of K, this does not appear to be the case.

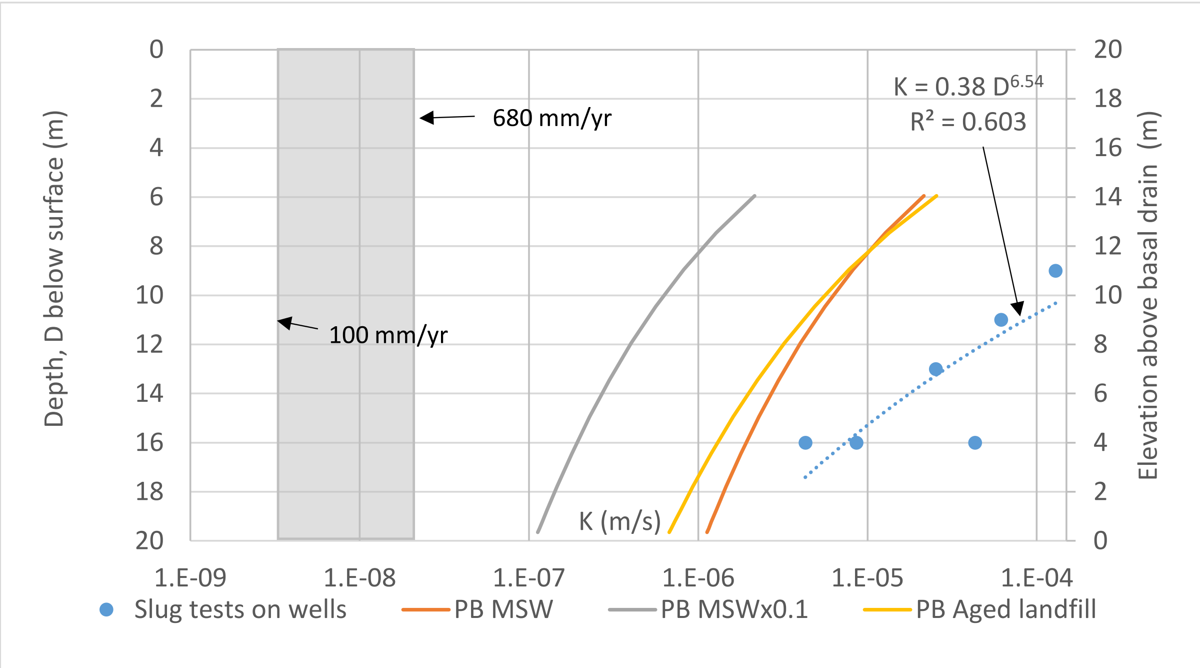

Figure 2 Variation of saturated hydraulic conductivity with landfill depth showing falling-head test results (K_h) and published values of K_v measured in fresh and aged MSW. Also shown is the range in infiltration rates at the site (between 100 and 700 mm/year, equivalent to $2.3x10^{-9}$ and $2.2x10^{-8}$ m/s). PB=Powrie Beaven 1999 K vs stress relationship

Numerical modelling

The aim of the modelling described in this paper is to explore potential causes of elevated pore water pressures in landfills. The most important functionality of any such model is the ability to model 2-phase flow (gas and leachate) within the context of a solid waste matrix whose hydraulic properties alter with effective stress. Consequently, the model chosen for the modelling exercise was LDAT (Landfill Degradation and Transport), which is a three-dimensional coupled gas and liquid flow model developed specifically for landfill applications. LDAT solves an array of landfill process constitutive equations using a finite difference algorithm (White eta l 2014) within a framework of rectangular representative elementary volumes (cells). As it is predominantly vertical liquid flow that is being investigated the model was run in one-dimensions. Full details of the model are given in White et al. (2003, 2004, 2013, 2014 and 2015).

Multi-component gas and liquid flow has been incorporated into the model allowing the determination of the detailed composition of gas and liquid fluxes, both at the boundaries and within the body of the model. The van Genuchten functions for relative gas and liquid permeabilities and capillary pressures are used to provide the coupling between the liquid and gas pore-pressure distributions and a constitutive flow equation based on Darcy's Law. For this paper, only the liquid-gas flow sub-model of LDAT was used, with gas generation being simulated as a constant rate gas injection into cells containing waste.

It is acknowledged that several major simplifications have been made within this modelling approach, including the one-dimensional nature of the modelling and the treatment of the waste as a single continuum. No account has been taken of potential preferential flow paths; both saturated and unsaturated flow have been assumed to take place within a single continuum. Evidence for preferential pathways in landfills is plentiful and at the detailed level, modelling of the flow processes using dual continuum gas and leachate models (e.g., Hu et al., 2020) may be required. However, the bulk hydraulic performance of the waste will be dominated by what happens in the matrix between preferential flow paths, and this is likely to be controlled by classical unsaturated flow behaviour. It is considered the modelling approach adopted here is a useful scoping / sensitivity study that provides insights into processes likely to occur in landfills.

Model output from LDAT is generated as .csv files with post processing and graphical analysis undertaken in Excel. An example of an excel post-processing data file is provided in supplementary information.

LDAT model configuration

A more comprehensive description of the LDAT setup is described in Supplementary Information and summarised as follows.

Grid and fixed boundary conditions

A one-dimensional vertical model was established to represent a 20 m thickness of waste overlying a 0.3 m thick gravel drainage layer. The model comprised 20 active elements (cells) with a surface area of 1m^2 stacked vertically. The lower 3 cells (each 0.1 m deep) represented the gravel and the remaining 17 cells (between 0.1 m and 1.5 m deep) the waste. A surface surcharge of 10 kPa was applied to represent a thin soil cover layer present at the site. Fixed gas and liquid pressures of 0 kPa were applied to the top boundary of the mesh. The gas and liquid pressures on the lower boundary were also normally set to 0 kPa (representing a fully drained gravel), but to test Hypothesis 2a the pressures on some simulations were set to 3 kPa to keep the gravel saturated. The gas and liquid permeabilities of the upper and lower boundary cells were set at values (gas 1 m/s; liquid 10 m/s) high enough to ensure there was no impediment to gas or leachate entering or leaving the model. A 'no-flow' condition was applied to the boundaries to the side of the active elements.

Material and unsaturated flow parameter values

LDAT adopts the empirical relationships between waste dry density / hydraulic conductivity and effective stress proposed by Powrie and Beaven (1999). These relationships are implemented in LDAT using the relative dry density (ρ_d) and relative saturated hydraulic conductivity (Ks) functions, defined below.

Dry density

$$\rho_d = \rho_{REF}\left(\frac{\sigma'}{\sigma'_{\rho REF}}\right)^{\gamma} \qquad (3)$$

Saturated hydraulic conductivity

$$K_s = fK_{REF}\left(\frac{\sigma'}{\sigma'_{KREF}}\right)^{\eta} \qquad (4)$$

The dry density is used to calculate the porosity of the waste as

$$\phi = 1 - \frac{\rho_d}{\rho_s} \qquad (5)$$

The values of the parameters in equations (3), (4) and (5) applied to the waste and gravel in the LDAT model setup are given in Table 1. The power law indices applied to the gravel result in a stiff material with a fixed hydraulic conductivity of 8.64 m/d (1×10^{-4} m/s). The Powrie and Beaven (1999) relationship for hydraulic conductivity of MSW was modified in the various model runs by multiplying by a fraction (*f*). This maintained the general form of the relationship between *K* and stress (i.e., that over a 20-metre landfill depth, *K* reduces by about two orders of magnitude) but introduced a necessary fitting parameter to the model. Although the exponent of the Powrie and Beaven (1999) relationship could have been altered to match the slope of the field data (Figure 2), it was considered that the short-term falling head tests undertaken, while acceptable for demonstrating a trend, were not comprehensive enough to justify such a change.

LDAT uses the van Genuchten unsaturated flow functions to calculate unsaturated liquid and gas permeabilities. The relative permeability function for the liquid phase was given in Equation 2. The equivalent relative permeability function of the gas phase is

$$K_{\Theta}^{G} = K_{S}^{G}(1-\Theta)^{l^G}\left[1-(\Theta)^{1/m}\right]^{2m} \qquad (6)$$

A more complete exploration of the relationship between liquid and gas relative permeabilities in unsaturated waste in LDAT is provided in White et al., (2014) and (2015).

Table 1 Physical material properties used in LDAT

Parameter		Notation	Units	Waste	Gravel
Dry density (Equation 3)	Reference dry density	ρ_{REF}	Kg/m^3	388	1534
	Reference effective stress	$\sigma'_{\rho REF}$	kPa	40	224
	Power law index	γ		0.248	0.025
	Solid phase particle density	ρ_s	Kg/m^3	1050	2650
Saturated hydraulic conductivity (Equation 4)	Reference K	K_{REF}	m/d	8.25	8.64
	Fraction of Reference K	f		Model dependent (see Table 3)	1
	Reference effective stress	σ'_{KREF}	kPa	40	40
	Power law index	η		-2.71	0
Water content	Initial degree of saturation		-	0.85	0.8

To cover the full potential range of unsaturated soil properties for waste, it was decided to investigate two sets (A and B) of van Genuchten (VG) properties (Table 2). Waste in all cells of the model was allocated parameters from either Set A or Set B, with no dynamic variation. Figure 3a and b show the water retention characteristic (WRC) and S_e versus relative *K* curves for these two sets of van Genuchten values: in terms of a sensitivity study, the parameters chosen largely cover the range of values for wastes found in other studies. Figure 3c relates the relative liquid permeability to the suction for the two different sets of VG property values.

Table 2 Mualem/van Genuchten unsaturated flow properties used in LDAT

Parameter		Notation	Units	Waste VG Set A	Waste VG Set B	Gravel
Liquid		α	1/kPa	3.47	0.2	2.45
		n	-	1.37	1.1	2.5
		m	-	0.27	0.09	0.6
	Residual degree of saturation	θ_r	-	0.2	0.2	0.03
	Maximum porosity saturation fraction	$\theta_s/\emptyset$	-	0.99	0.99	0.99
	Correlation index	$l^L = l^G$	-	0.5	0.5	0.5
	Reference			This paper	This paper	Mace et al., (1998)

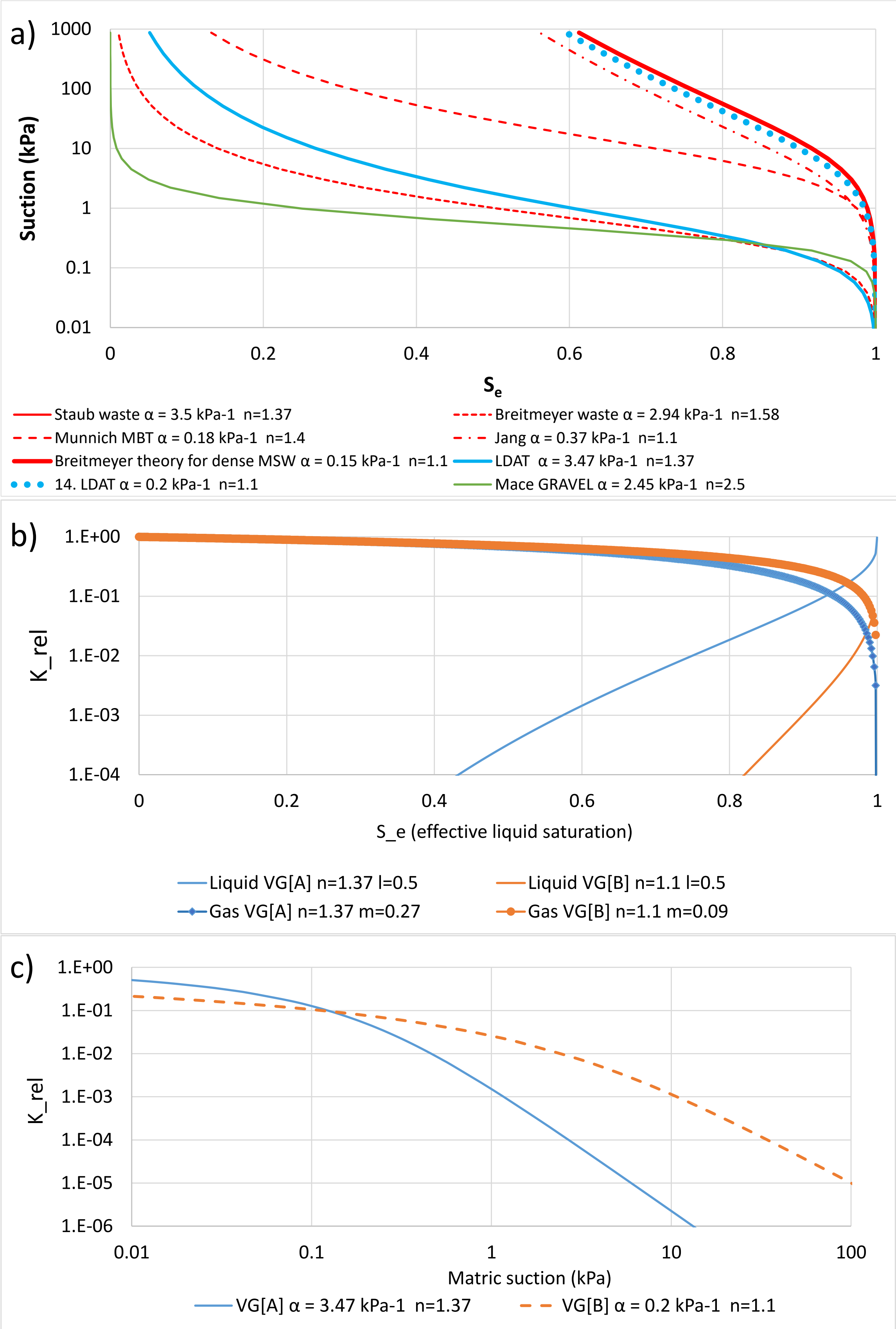

*Figure 3 Plots of unsaturated flow properties of wastes modelled in this paper **a)** WRC of van Genuchten (VG) parameter Set [A] and Set [B] compared to other waste materials; **b)** Relative liquid and relative gas permeability plotted against degree of liquid saturation S_e for VG Set [A] and VG Set [B]; **c)** Relative liquid permeability plotted against suction for VG Set [A] and VG Set [B]*

Importantly, although Figure 3b shows that, for a given Se, the relative liquid permeability for VG[B] is less than for VG[A], for a given suction above 0.1 kPa the relative permeability for VG[A] is less than for VG[B] (Figure 3c). This is because at a given suction (say 10 kPa) wastes with properties VG[A] will have a S_e of ~0.3 (Figure 3a) while there will be relatively little desaturation (S_e ~0.9) of wastes with properties VG[B].

Infiltration and gas generation inputs

The initial water content of all models was set as the degree of saturation, with values of 0.98 (i.e., near fully-saturated conditions) occurring from the base to an elevation of 12.5m, then dropping to 0.85 at top of the waste profile. These values were chosen to aid model stability and convergence rather than attempting to represent any actual conditions at the site.

Vertical infiltration was applied as a constant flux into the upper boundary cell. Flow leaving the model collects in the bottom boundary cell. Most model simulations were run with an infiltration rate of 400mm/year to replicate conditions at the site. An infiltration rate of 50 mm/year was also simulated to reflect modern containment landfills which are often capped with a low permeability top cover to restrict infiltration to 50 mm/year or less.

Steady state gassing was introduced as a source term (G) into each of the waste cells at a rate proportional to the height of the cell. Two different gassing rates were initially investigated. The higher rate of 0.37 kg_{LFG} /day (5.63 $m^3_{LFG}\ t_w^{-1}\ yr^{-1}$) was similar to the rate of 6.2 $m^3_{LFG}\ t_w^{-1}\ yr^{-1}$ used by Thiel (1999) (also reported by Merry et al., (2006)) for normal decomposition of landfills. A lower rate 0.04 kg_{LFG} /day, approximately equivalent to the gas generation rate of 0.61 $m^3_{LFG}\ t_w^{-1}\ yr^{-1}$ at the site was also modelled (**Table 3**). Some limited runs were also undertaken at rates of 0.07 kg_{LFG}/day (1.06 $m^3_{LFG}\ t_w^{-1}\ yr^{-1}$) and 2.4 kg_{LFG} /day (36.5 $m^3_{LFG}\ t_w^{-1}\ yr^{-1}$; an upper rate for bioreactor landfills with enhanced decomposition, (Thiel, 1999).

Model runs

LDAT models transient conditions. However, all models were run for a long enough time for pseudo steady state conditions to become established (i.e., there was a <5% difference between input and output fluxes, and stable pressure profiles). Significant model runs are summarised in **Table 3**.

Results

Models investigating a lower than expected waste hydraulic conductivity (Hypothesis 1)

In the absence of landfill gas production, and using the original Powrie-Beaven (PB) 1999 relationship between hydraulic conductivity and effective stress (i.e., f = 1), there is no build-up of any positive pore-water pressures or leachate heads in the model (for example, simulation M41a for f = 0.1 :Figure 4a) with an infiltration rate of 400 mm/yr. The waste remains under suction throughout its full depth although there is a difference between the suction profiles developed as a consequence of the differing van Genuchten parameter sets chosen (Figure 4b). It is not until f is reduced to 0.002 (M41ab, Figure 4a) that a reasonable match between the modelled and observed leachate pressures occurs. As to be expected, the pore water pressures curves for a given value of f for VG[A] and VG[B] (not shown) overlie each other exactly within the saturated part of the profile (i.e., below an elevation of ~ 14 m in the case of f = 0.002, Figure 4c).

Model Name	Description/ other conditions	Permeability f (Eq. 4)	Lower boundary pressure (kPa)	Infiltration R (mm/yr)	Gassing rate $G^{\$}$ ($m^3 t_w^{-1} yr^{-1}$)	Gassing rate $G^{\#}$ (kg/day)	VG parameter set for waste	Fig. number where results displayed	Approx' convergence time (model days)	Testing Hypothesis
M41a M41b	Baseline model with f=0.1	0.1	0	400	None	None	A B	4a,4b,6 4b	600 550	1
M41aa M41ab M41g	Baseline model where f altered to match leachate heads/ pressures	0.001 0.002 0.003	0	400	None	None	A	4a 4a, 4c 4a	2,500 1,600 1,100	1
M31f M31h M31q M31p	K_{sat} of 0.1m layer directly above gravel fixed at $1x10^{-10}$ m/s	0.1 0.03 0.003 0.002	0	400	None	None	A	5 5 5 5	500 3,000 3,000 2,500	2a
M41k	Lower gravel kept saturated	0.003	3	400	None	None	A	-	2,500	2b
M41L	Lower gravel kept saturated	0.003	3	400	None	None	B	-	2,500	2b
M42a	Low gassing rate with baseline	0.1	0	400	1.06	0.07	A	9	800	3
M42b	Low gassing rate with baseline	0.1	0	400	1.06	0.07	B	9	600	3
M43a	Moderate gassing rate with baseline	0.1	0	400	5.6	0.37	A	6,9	500	3
M43b	Moderate gassing rate with baseline	0.1	0	400	5.6	0.37	B	6,9	720	3
M46ab	Low gassing rate (site conditions) with baseline	0.1	0	400	0.61	0.04	A	9	600	3
M46c	Low gassing rate (site conditions) with baseline	0.1	0	400	0.61	0.04	B	9	600	3
M47a	High gassing rate with baseline	0.1	0	400	36.5	2.4	A	9	500	3
M47b	High gassing rate with baseline	0.1	0	400	36.5	2.4	B	9	700	3
M46ae M46af* M46ad	Landfill gassing rate at study site	0.005 0.006* 0.008	0	400	0.61	0.04	A	7a 7a, 8 7a	1,000	3
M46b M46bd* M46bb	Landfill gassing rate at study site	0.01 0.015* 0.02	0	400	0.61	0.04	B	7b 7b, 8 7b	1,300	3
M48a	Low infiltration, moderate gassing rate R/f=5	0.01	0	50	5.6	0.37	A	-	4,000	3
M48ca	Low infiltration, high gassing rate R/f=5	0.05	0	50	36.5	2.4	A	10a, b	~5,000	3
M48e	Low infiltration, high gassing rate R/f=5	0.05	0	50	36.5	2.4	B	10c, d	~5,000	3

* Best fit to field data $^{\$}$ t_w=tonne of waste $^{\#}$ Since the cross-sectional area of the LDAT one dimensional model was $1m^2$, these units are equivalent to kg m^{-2} day^{-1}

Table 3 Summary of LDAT model runs

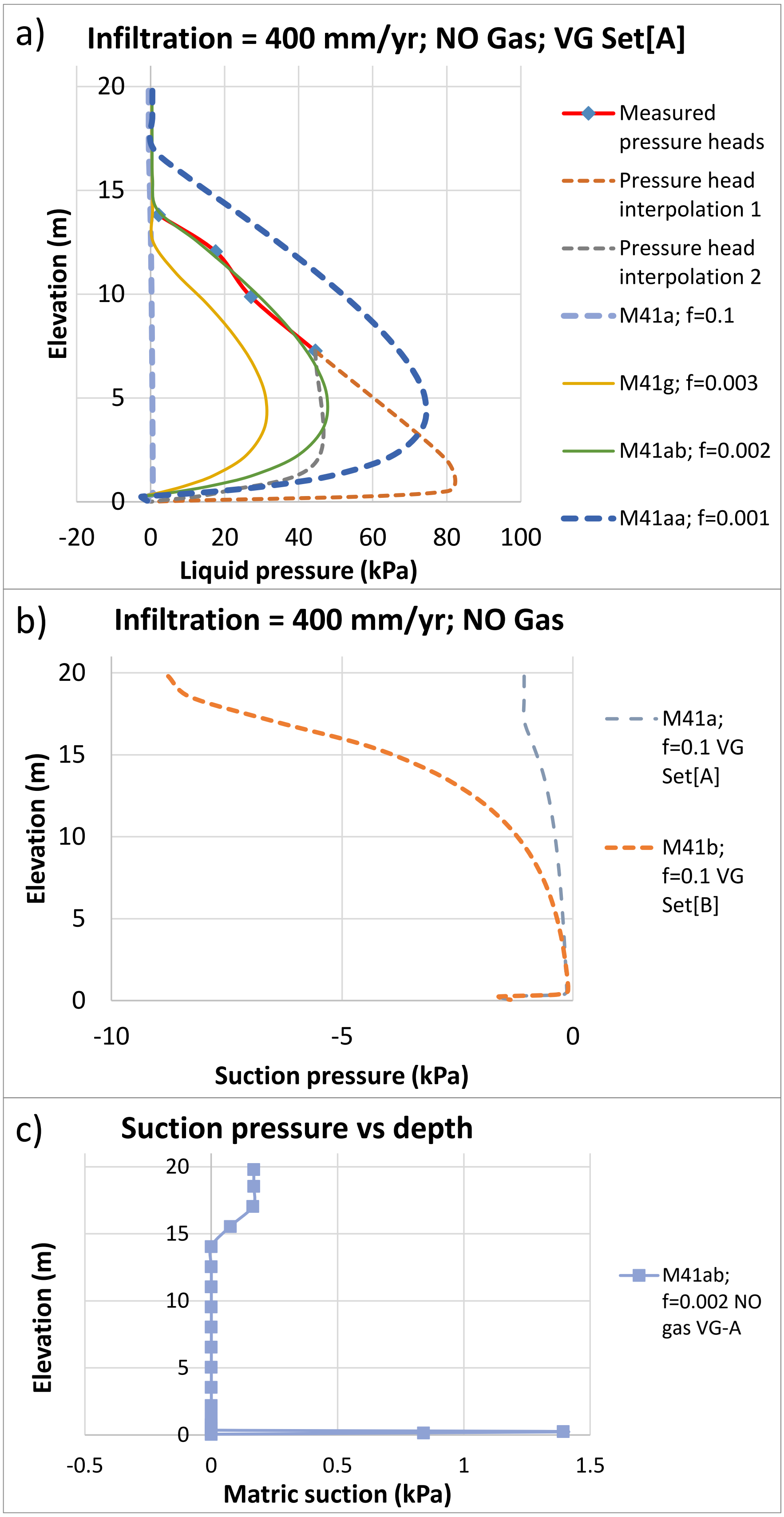

*Figure 4 **a)** Relationship between f and pore water pressure in 20 m deep non-gassing landfill with 400 mm/yr infiltration and fully drained basal gravel ; **b)** Suction pressure profile for f=0.1 for van Genuchten parameter sets A and B; **c)** suction pressure profile for f=0.002*

Models investigating a low permeability band at the waste/gravel interface (Hypothesis 2)

The simulation of a low permeability band just above the gravel interface – representing, for example, clogging – combined with a waste permeability factor *f* > 0.03 (M31f & M31p), results in a hydrostatic pressure profile above the interface (Figure 5), and potentially creates a better match with the pressure head interpolation 1. This would imply that the low permeability band is the dominant control over head distribution in the site. However, the uniform gradient of this profile only matches the observed leachate heads near the water table (the top of the saturated waste). Combining a low permeability interface with a waste permeability factor *f* = 0.003 (M31h) results in a very different pore water pressure profile near the gravel compared with the case without the low permeability band (M41g; Figure 5). It also results in an overall increase in pore water pressures throughout the profile and a rise in the level below which the waste is fully saturated to ~16m. A combination of a low permeability zone above the gravel and different values of *f* cannot reproduce the s-shape of the measured pressure heads in the site.

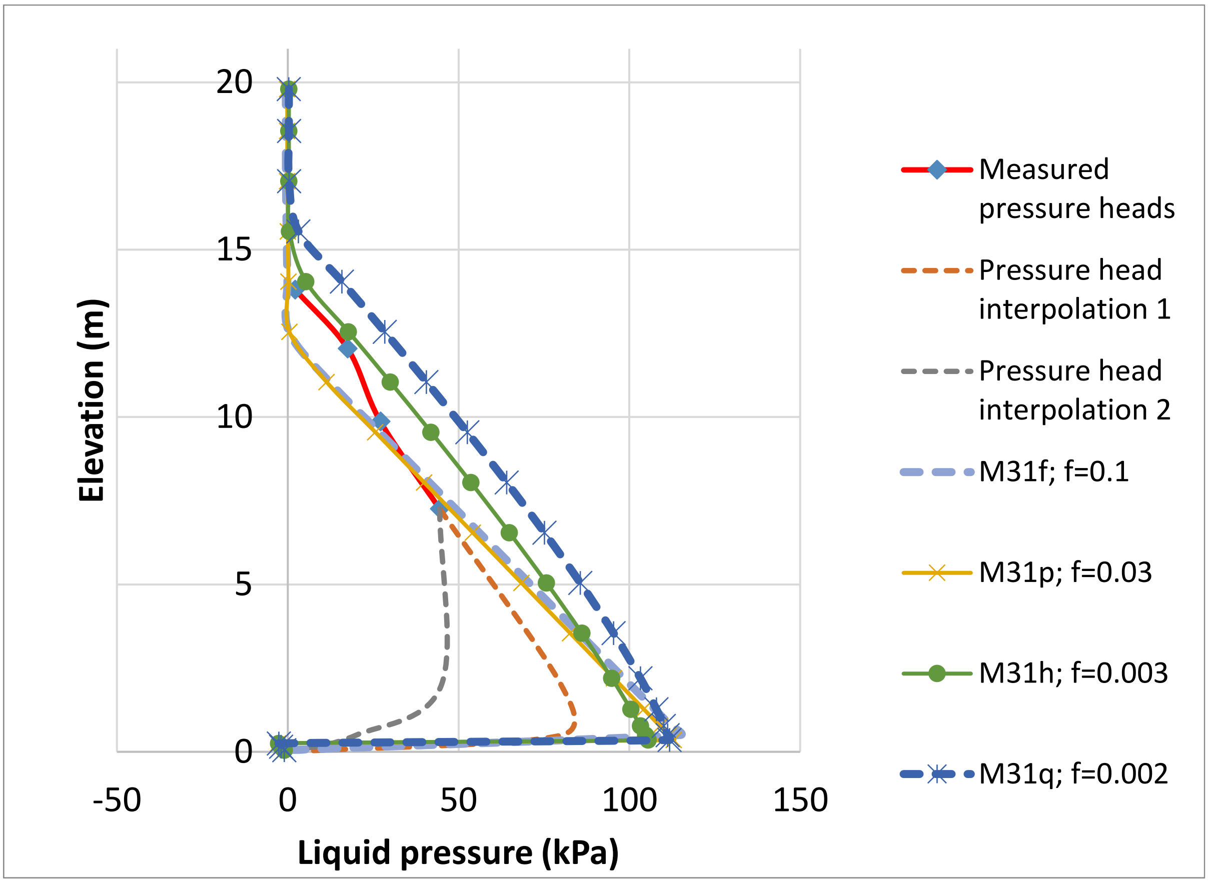

Figure 5 Relationship between f and pore water pressure in a 20 m deep non-gassing landfill with interface clogging, 400 mm/yr infiltration and fully drained basal gravel

Hypothesis 2b speculated that in the absence of a physical low permeability zone, fully draining the gravel layer might result in the partial dewatering of the waste directly above it hence causing a localised reduction in relative permeability. Simulation M41ab (Figure 4) investigates this for VG[A]. For a direct comparison, simulations were run where the gravel was kept fully saturated with a porewater pressure at the top of the gravel of 0 kPa. Other than increased pore water pressures in the gravel, these gravel saturated runs gave exactly the same pore water pressure profile within the waste (hence the results are not shown) as when the gravel was fully drained, irrespective of the van Genuchten parameter set used.

Models investigating gassing (Hypothesis 3)

Results from analyses simulating gas generation are shown in Figure 6, Figure 7 and Figure 9

The introduction of *in situ* gas generation into the models resulted in a marked increase in simulated pore water pressures. The impact of the different VG parameter sets is also clear (Figure 6). Simulations M43a and M43b, with *f* = 0.1 and gassing at 5.6 $m^3_{LFG}\,t_w^{-1}\,yr^{-1}$ (0.37 kg/day), can be compared with results for M41a (also shown on Figure 4), where there was no build-up of any pore water pressures in the absence of gassing. With VG parameters Set A, the maximum pore water pressure in M43a was ~5 kPa, but this increased by a factor of about 4 (to 20 kPa) in model M43b, in which VG parameters Set B were used.

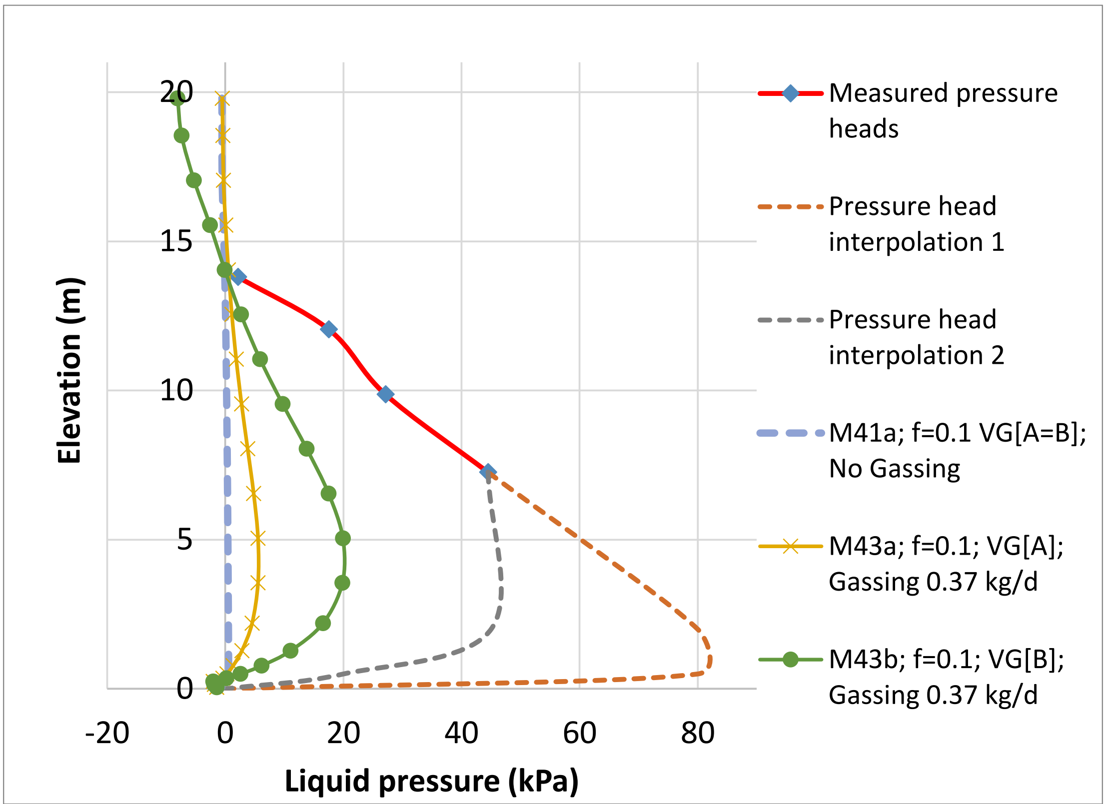

Figure 6 Relationship between f and pwp in 20 m deep landfill gassing at 0.37 kg/d (5.6 $m^3_{LFG}\,t_w^{-1}\,yr^{-1}$), 400 mm/yr infiltration and fully drained basal gravel

Applying the more site-specific gas generation rate of 0.61 $m^3_{LFG}\,t_w^{-1}\,yr^{-1}$ (0.04 kg/day) required the value of *f* to be reduced considerably to give pore water pressures that matched the values measured on site. With VG parameters Set A, a value of *f* = 0.006 was required (Figure 7a), compared with *f* = 0.002 with no gas (Figure 4). With VG parameters set B, the required value of *f* was ~0.015 (Figure 7b). Thus, introducing a modest amount of gas into the model resulted in *f* having to be increased by a factor of between 3 (VG[A]) and 7.5 (VG[B]) to provide an approximate match to pore water pressures, compared with the no-gassing scenario.

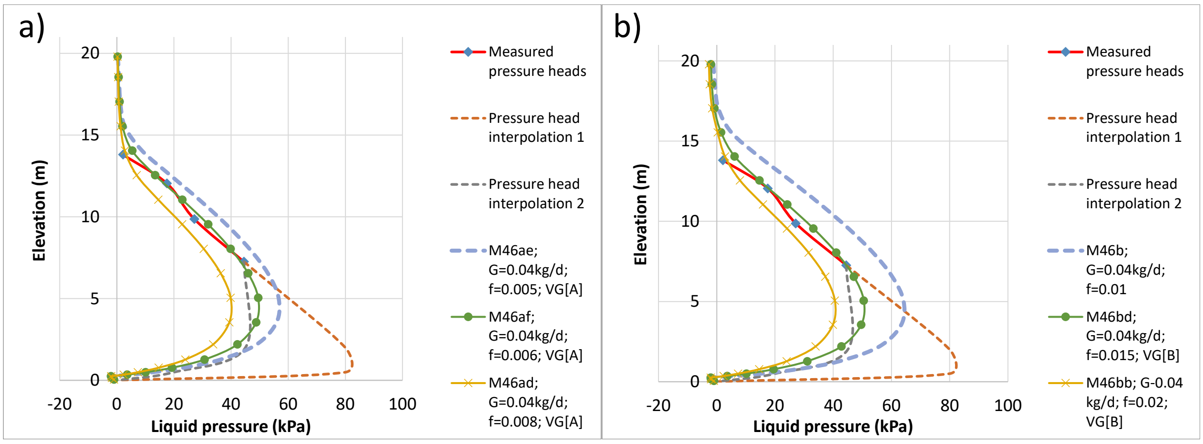

*Figure 7 Relationship between f and pwp in 20m deep landfill gassing at 0.04 kg/d (0.61 $m^3_{LFG}\, t_w^{-1}\, y^{-1}$), 400 mm/yr infiltration and fully drained basal gravel for **a)** VG Set [A] and **b)** VG Set [B}*

Figure 8 shows that the unsaturated profile of hydraulic conductivity for both VG parameters Set A and B are the same and that the hydraulic conductivity becomes less than the average infiltration rate below 5 metres above the base of the site. The saturated hydraulic conductivity (which is entirely theoretical because gassing prevents saturation occurring) in Figure 8 shows a similar shaped profile to that consistent with the porewater pressure profile demonstrated by Powrie and Beaven (1999) for 1-D downward flow in a landfill where hydraulic conductivity is controlled by effective stress. The adoption of VG[B] results in a one order of magnitude reduction in the hydraulic conductivity (compared with the saturated value) throughout the lower part of the depth profile, while for VG[A] the hydraulic conductivity in the lower part of the waste profile is reduced by a factor of approximately 4.

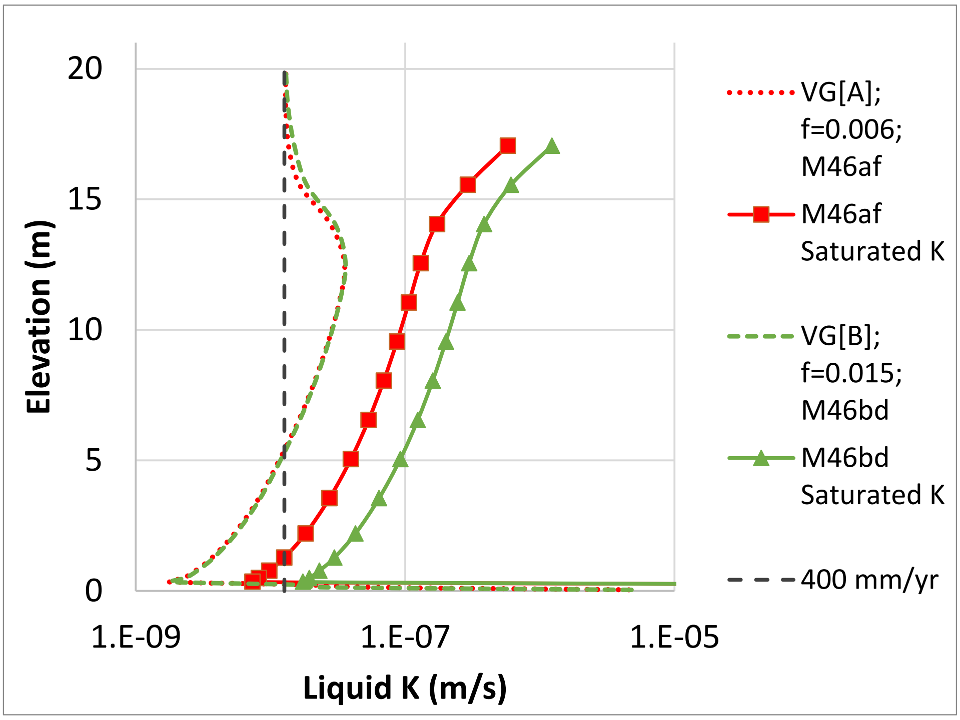

Figure 8 Relationship between unsaturated hydraulic conductivity and theoretical saturated K showing impact of gassing for different van Genuchten properties of waste; 400 mm/yr infiltration and fully drained basal gravel

In all the above cases where both the upper and lower boundaries were open to gas flow, approximately three times as much gas left the model through the upper boundary as through the lower.

Plots of the maximum pore water pressure within the waste profile, and the depth from the surface to the water table (as defined by pore water pressure > 1kPa) are shown in Figure 9 for different gas generation rates, assuming a vertical infiltration rate of 400 mm/year and *f*=0.1 (i.e., with conditions considerably more permeable than at the study site). The no-gas scenarios (simulations M41a and M41b, Figure 4a and b) do not result in any build of pore water pressures; and there is no water table within the landfill, as the minimum vertical saturated permeability of the waste (~1.4x10^{-7} m/s)) is at least 1 order of magnitude greater than the infiltration rate of 1.27x10^{-8} m/s (Figure 2). However, introducing even modest gas generation rates into the model results (Figure 9) in a significant depth of waste where pore water pressures are in excess of 1 kPa (10 cm water head).

Models investigating low infiltration landfills

Figure 10 shows the results of simulations where the infiltration rate was 50 mm/year, and gassing was 36.5 $m^3_{LFG}\, t_w^{-1}\, yr^{-1}$. Figure 10 a) and b) show simulated gas and liquid pressures and the degree of saturation for a landfill with VG parameter set A, whilst Figure 10 c) and d) are for a landfill with VG parameter set B. The implications of these results are explored further in the discussion section.

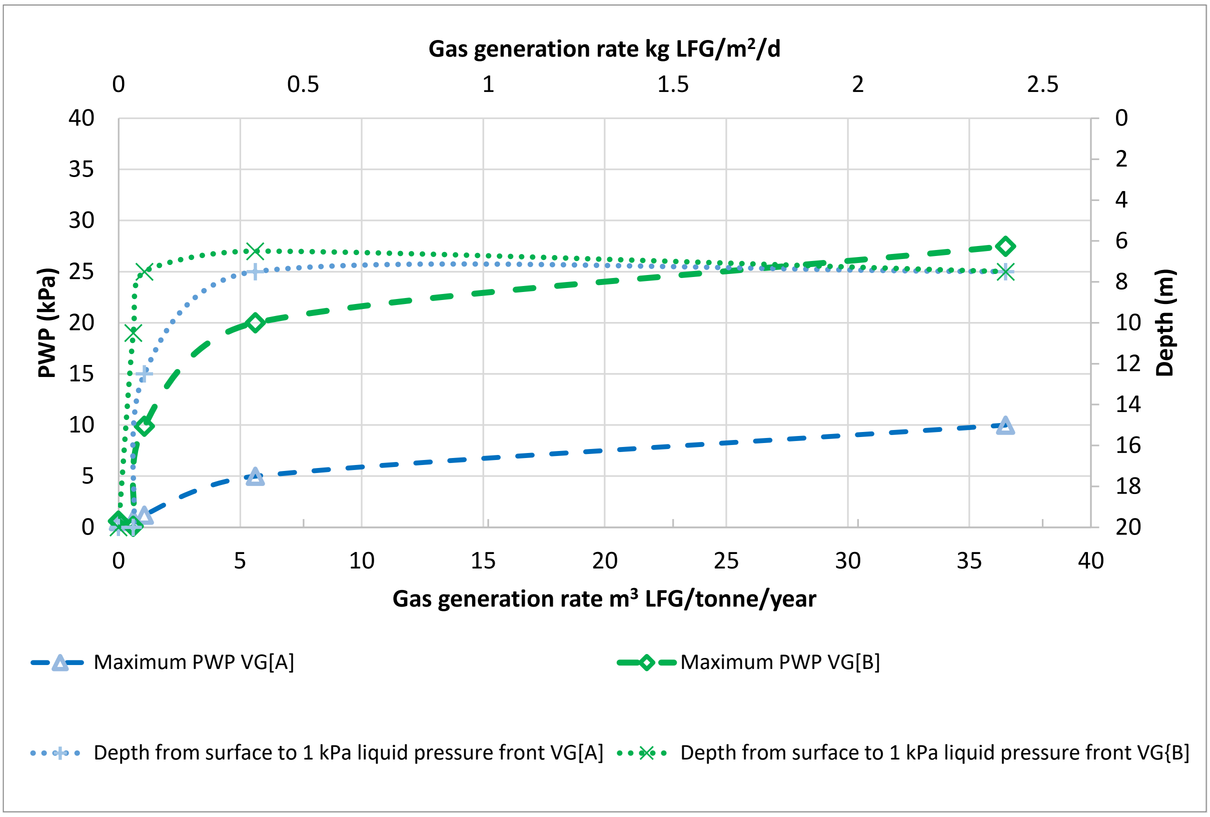

Figure 9 Maximum porewater pressure (PWP) and depth from surface at which water table conditions (defined as PWP>1 kPa) are reached in a 20 m waste profile with different gassing rates for van Genuchten parameter Set [A] and Set [B]. Infiltration = 400 mm/yr, f=0.1.

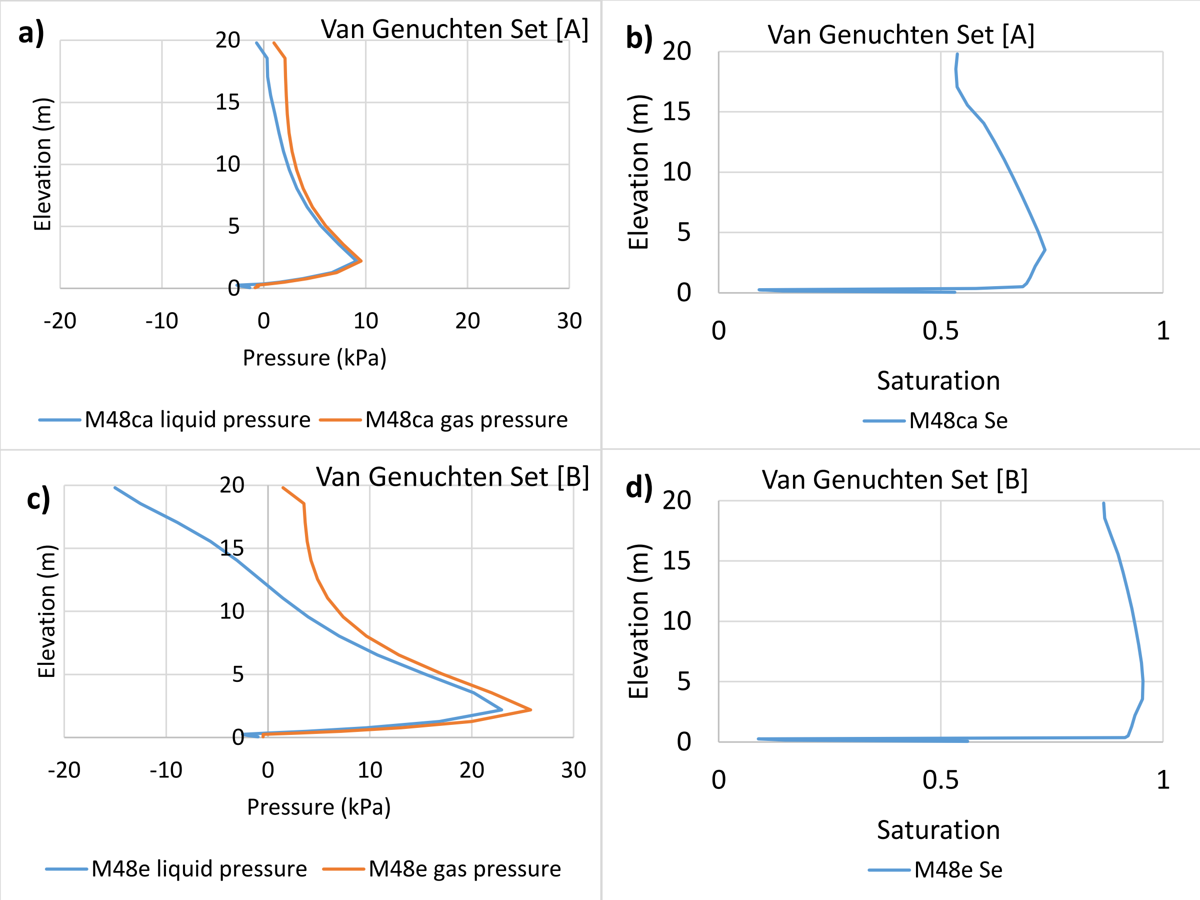

Figure 10 Se and pore water pressure at the gravel/ waste interface in a 20 m deep non-gassing landfill with van Genuchten parameter Set [A], 400 mm/yr infiltration and fully drained basal gravel

Discussion

The aim of the modelling described in this paper was to explore potential causes of elevated pore water pressures in landfills. To reduce the number of variables investigated, the Powrie-Beaven relationship between *K* and effective stress was fixed in the model. This is a reasonable assumption as although different waste types will have individual correlations between hydraulic conductivity and density variations between the logarithm of the hydraulic conductivity and vertical effective stress are much less pronounced (Beaven et al., 2008).

In the case study discussed, there are major discrepancies between the hydraulic conductivities measured in the falling head tests (presumed to characterise predominantly K_h) and the values of K_v that the modelling indicates are required to generate the measured pore water pressure profiles, with values up to ~40 kPa. Although the modelling is not capable of identifying specific causes, anisotropy in hydraulic conductivity is one very likely reason. In addition to anisotropy in the wastes, landfill-scale anisotropy could be caused by the presence of any number of low permeability layers (for example, daily cover) that, even if not continuous across the whole site, are extensive enough to disrupt flow paths very significantly. In the absence of landfill gas generation, a reduction in a baseline relationship between hydraulic conductivity and effective stress (Powrie & Beaven, 1999) by a factor of approximately 500 (*f* ~ 0.002) was required to replicate the pore water pressure profile observed in the site, for an infiltration rate *R* = 0.4 m/year.

Based on the monitored pore water pressures in the landfill, localised clogging at the gravel to waste interface (for example, caused if a geotextile had been used) can be ruled out as a mechanism which alone could cause the elevated pore pressures at this site. If interface clogging was present, it would need to be operating in conjunction with another mechanism to reduce the bulk hydraulic conductivity of the waste. Measurements of pore water pressure in the waste directly (ideally within 1 m) above the gravel drainage layer are required to provide better insights into whether interface clogging is occurring at this and other landfill sites.

The model results indicate that whether a gravel leachate drainage layer is operated fully drained or fully saturated (0 kPa pressure at top) should have no impact on pore water pressures in the overlying waste *in the absence of gassing*. A further exploration of this finding is provided in supplementary information.

In situ landfill gas generation has a major impact on increasing pore water pressures (by changing the degree of saturation and the permeability to water), and these pressures are strongly influenced by choice of the van Genuchten parameters for the waste (see Figure 3). To fit the model to the case study field data with the actual landfill gas generation rate of 0.61 $m^3_{LFG}\,t_w^{-1}\,yr^{-1}$ and an infiltration rate of 400 mm/yr, the adopted permeability relationship had to be reduced by a factor of between ~170 (M46af; *f* = 0.006) for VG[A] and ~70 (M46bd; *f* = 0.015) for VG[B]. This is because gassing at a given rate causes a greater reduction in the unsaturated hydraulic conductivity in wastes with van Genuchten properties closer to Set B than Set A. Combining these values of *f* with the observation that the Powrie-Beaven relationship for K_v was already approximately an order of magnitude lower than the measured K_h values, leads to a tentative conclusion that landfill scale (rather than waste scale) anisotropy could be as high as between 1:700 to 1:1500 at the study site.

The results of this modelling exercise provide an explanation of how an open tipped piezometer installed into nominally unsaturated gassing waste (s_e<1) could provide readings indicating free draining leachate within the landfill. The example of a 20 metre deep landfill with an infiltration rate of 50mm/year is used to illustrate this point.

Figure 10a shows simulated (M48ca) gas and liquid porewater pressures in excess of 1 kPa (10 cm water head) within the full landfill depth profile (with VG parameters Set A, f=0.05, R=0.05m/yr and a gas generation rate of 36.5 $m^3_{LFG}\,t_w^{-1}\,yr^{-1}$), while the degree of saturation s_e (Figure 10b) varies from ~0.55 to 0.75. Based on simulation M48ca, an open tipped piezometer installed into the waste mass, at a depth of (say) 5 m (equivalent to 15 m elevation), would encounter a gas pressure of 2.3 kPa and a slightly lower liquid pressure of 1.1 kPa (Figure 10a). As both these pressures are in excess of the 0 kPa associated with an empty piezometer pipe there would be an initial potential for both gas and liquid to enter the piezometer even though the degree of saturation of the waste is less than 70%. If the piezometer tube is left open to the atmosphere then, over time, the tube will fill up with a leachate column of 0.108 m. While the tube is filling up with leachate, gas pressures will be in excess of water pressures and there will be a release of gas bubbling through the leachate column to the atmosphere. Adopting VG set B ((simulation M48e); Figure 10c and d) results in a very different response at an assumed piezometers installed into the waste. Negative pore water pressures (suctions) exist within the upper 8 m of the waste profile such that there would be no potential for any liquid to enter a piezometer, although landfill gas would readily enter the piezometer (positive gas pressures). Below about 8 m depth, both leachate and gas would enter a piezometer even though the waste is not fully saturated (s_e <1).

Lower rates of landfill gas generation at this infiltration rates can also result in elevated porewater pressures in the waste column depending on the value of *f*. As an example, simulation M48a (results not shown) for a 20m deep landfill with an infiltration rate of 50 mm/ year, f=0.01, VG Set [A] and a gas generation rate of 5.6 m^3_{LFG} t_w^{-1} yr^{-1} (0.37 kg/day) causes maximum porewater pressures of 15 kPa and water table type conditions to within 3 m of the surface.

Increased landfill gas generation rates lead to increased maximum pore water pressures, but this relationship is not linear (e.g., Figure 9). Gas generation in wastes with van Genuchten parameters closer to set B (i.e., low values of n) has the largest impact on increasing maximum pore water pressures for a given value of *f*. However, wastes with van Genuchten parameters closer to set A result in water table conditions over a larger proportion of the landfill depth.

This modelling exercise has demonstrated the importance of both landfill gas generation and anisotropy of hydraulic conductivity (the K_h to K_v ratio) within the landfilled waste in the development of pore water pressures in vertically draining landfills. It is likely that very large landfill-scale anisotropy combined with gassing is causing the bulk vertical hydraulic conductivity to reduce to below the infiltration rate.

The studied landfill is a fully lined above-ground land-raise that raises the likelihood of lateral flow into seepage faces on the flanks of the site. Consequently, there is a need to expand the modelling to two dimensions to take this into account, especially as anisotropy will favour horizontal flow over vertical.

The simulations that most accurately reflected the porewater profiles and boundary conditions at the site (for example, simulations M46af and M46bd, **Table 3**), took many years in terms of model time to reach equilibrium (i.e., good comparison between input and output fluxes and stable pressure profiles). This is caused by a combination of high starting water content conditions, relatively low infiltration rates and low waste effective permeability meaning that changes in the water content of the waste column are slow. As noted above the single continuum modelling approach used does not account for preferential pathways, but the conclusion that bulk changes to water content are likely to take a considerable time are considered valid.

Conclusions

A one-dimensional two-phase numerical model of leachate and gas, LDAT, has provided insights into the possible causes of high leachate heads in landfills underlain by a fully functioning leachate collection system. Model results were compared with porewater pressure profiles in an above-ground restored landfill with leachate recirculation and a fully drained basal leachate drainage system. Vertical drainage to the basal drainage layer was estimated as 400 mm/year and landfill gas generation approximately 0.61 m^3 LFG $t^{-1}yr^{-1}$.

Falling head tests in piezometers at the landfill indicated waste hydraulic conductivities approximately 10 times greater than an empirical relationship between effective stress and permeability demonstrated by Powrie and Beaven (1999). However, Powrie and Beaven's relationship was based on vertical hydraulic conductivity measurements whereas falling head tests are dominated by horizontal hydraulic conductivity, and this difference could be accounted for by waste anisotropy.

A review of the available literature on the unsaturated properties of waste resulted in the proposition that the unsaturated properties of waste can be bounded by two sets of van Genuchten parameters. One end member of these van Genuchten parameters (Set A) had similarities to freely draining naturally occurring materials, such as sands where the material can desaturate very substantially before large suctions are generated. The other end member (Set B) had similarities to naturally occurring materials such as silty or clay loams, where large suctions can develop with a relatively small amount of desaturation.

In the absence of any gassing within the waste, the measured pore water profile within the landfill could be simulated if the Powrie-Beaven relationship between hydraulic conductivity and vertical effective stress was reduced by a factor of 500 (f = 0.002). Simulation of a low permeability interface between the basal drainage layer and the overlying waste ruled out such an interface as being the only cause of the elevated porewater pressures within the landfill. It was also not possible to reproduce the shape of the measured pressure heads in the site by combining a low permeability interface with different waste permeability factors (f).

The model results indicate that whether a gravel leachate drainage layer is operated fully drained or fully saturated (0 kPa pressure at top) should have no impact on pore water pressures in the overlying waste *in the absence of gassing*.

The generation of landfill gas within nominally saturated waste causes the waste to partly desaturate, the effect of which is strongly influenced by the chosen unsaturated VG parameters of the waste.

Increased landfill gas generation rates lead to increased maximum pore water pressures, but the relationship is not linear. Gas generation in wastes with van Genuchten parameters closer to set B (i.e., low values of n) has the largest impact on increasing maximum pore water pressures for a given value of f.

Introducing a distributed gas generation source term based on the landfill's estimated actual gas generation rate of 0.61 m^3 LFG $t^{-1}yr^{-1}$ changed the value of f required to create a fit with the observed pressure head profile in the waste. The adopted permeability relationship had to be increased by a factor of between ~3 (f = 0.006 for VG[A]) and ~7.5 (f = 0.015 for VG[B]) from the no-gas scenario (f=0.002). This is because gassing at a given rate causes a greater reduction in unsaturated hydraulic conductivity in wastes with van Genuchten properties closer to Set B than Set A.

Combining the values of K_v derived from the numerical modelling with the measured K_h values at the site, leads to a tentative conclusion that landfill scale anisotropy could be as high as ~ 1:1000 at the study site (i.e., K_h is approximately 3 orders of magnitude higher than K_v).

Introducing even modest gas generation rates into the model can result in a significant depth of waste where porewater pressures are in excess of 1 kPa (10 cm water head). This results in apparent below water table type conditions as water will enter piezometers installed into such wastes, even though the gassing reduces the degree of saturation to below 1. Wastes with van Genuchten parameters closer to set A result in water table conditions over a larger proportion of the landfill depth than set B.

The studied landfill is a fully lined above-ground land-raise that increases the likelihood of lateral flow into seepage faces on the flanks of the site. Consequently, there is a need to expand the modelling to two dimensions to take this into account, especially as anisotropy will favour horizontal flow over vertical.

Acknowledgments

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