Accelerated flushing of contaminants from MSW landfill at the field scale using a fill-and-draw method

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ABSTRACT

A fill-and-draw flushing test on a landfill cell containing MSW waste was carried out to examine the operational viability of this method for accelerating the flushing of contaminants and landfill stabilisation.

During the fill cycle, 800 m^3 of water containing the tracer bromide was pumped into the base of a 0.44 ha landfill cell, resulting in the estimated saturation of 9,400 m^3 of waste. Abstraction took place in two phases, during which 1,100 m^3 of tracer/leachate was recovered. Samples of leachate were analysed for the tracer, electrical conductivity and indigenous solutes chloride and ammonia. Tracer recovery was between 63 and 72% for bromide. An estimated 227 kg of ammonia and 575 kg of chloride were removed.

Test data was used to calibrate a 1D, dual-porosity model involving advection in a mobile zone, and diffusion into 'blocks' of a less mobile zone. The model fitted well to the early time data, whereas later data appears to have been affected by recharge.

The results of this trial demonstrate the possibilities of the 'fill-and-draw' concept using the basal leachate drainage system of landfills as a potential accelerated landfill remediation technique. However, modelling results suggest low contaminant removal efficiency. Including a pause between the fill and the draw cycles improves mass removal.

1 INTRODUCTION

The storage of wastes in engineered low-permeability containment landfills can significantly reduce the risk of environmental contamination (Scharff, 2014). However, by sealing the waste in this manner, the pollution potential of the waste itself is not rapidly reduced, and the risk of an uncontrolled release of pollutants remains over a timescale that is often measured in centuries (Knox, 1990). Therefore, active treatment, monitoring and maintenance may be required which may pose a considerable, long term financial cost (Beaven et al., 2014).

Fill-and-draw (alternatively known as flood-and-drain) is one potential method for accelerating the flushing of indigenous contaminants from landfill. One application of the technique would involve saturating *in situ* waste from the bottom up with water and then draining the landfill to recover soluble and diffusive contaminants. The success of any contaminant flushing technique will be dependent on the contaminant transport properties of the waste. Factors linked to contaminant transfer include, the waste type, its geochemical and geotechnical properties, heterogeneity over a variety of scales and the duration of the flushing process (Beaven et al., 2005). The flushing method may have to be repeated a number of times to significantly reduce the contaminant load (Bolyard & Reinhart, 2016). There is a need to establish the efficacy of this method versus other means of accelerating contaminant removal.

This paper describes a fill-and-draw tracer test carried out in a large-scale pilot landfill with the aim of recovering contaminant transport properties of saturated waste at this large scale. The tracer bromide (Br, in the form of potassium bromide) was used in the experiment. The tracer was mixed with water and injected into the landfill. Water has a naturally low concentration of dissolved solutes compared to that typically found in leachate, therefore, electrical conductivity (EC) was also measured as an indicator of contaminant removal. The dilution of indigenous contaminants ammonia and chloride by the injected water, provided a further opportunity to study contaminant transport behaviour in comparison with the tracers. Test data are analysed using a simple one-dimensional (vertical) dual-porosity model. Recovered transport parameters are subsequently used to simulate various flushing regimes to explore the best operational practice for the fill-and-draw method and long-term contaminant removal.

2 BACKGROUND

One method of accelerating landfill completion is through enhanced flushing of fluids. This will encourage both degradation and the removal of soluble contaminants from the waste (e.g. Beaven et al., 2014, Bolyard & Reinhart, 2016). Research has shown that, within waste, dual-porosity flow will dominate (e.g. Rosqvist & Destouni, 2000, Woodman, 2007, Woodman et al., 2014). This dual-porosity structure facilitates preferential or by-pass flow and short-circuiting through inter-connected macro-pores, termed the mobile zone, that surround immobile zones or 'blocks' of lower permeability waste where advective flow is negligible. Preferential flow will leave areas of waste unaffected by or with limited flushing. This flow behaviour has been reported in several field studies (Harris, 1979; Ehrig, 1983; Blight et al., 1992; Bendz et al., 1997; Rosqvist et al., 1997), and in laboratory studies (e.g., Blakey, 1992; Zeiss and Uguccioni, 1997).

Where dual-porosity flow dominates, solute dispersion in the liquid phase will be dominated by diffusive exchange, with mechanical dispersion being relatively unimportant (Bendz and Singh, 1999; Rosqvist & Destouni 2000; Beaven et al., 2003; Döberl et al., 2003; Fellner et al., 2003; Woodman, 2007; Fellner et al., 2009; Hu et al., 2020; Rees-White, 2021). The relative contribution of these processes will vary between sites and under different fluid flow conditions. In saturated conditions, dual-porosity contaminant exchange has been demonstrated in column studies (e.g. Rosqvist & Bendz, 1997; Beaven et al., 2003; Fellner & Brunner, 2010, Woodman et al., 2014), and in field-scale tests (Woodman et al., 2017; Rees-White et al., 2021). Horizontal solute transport within a 17 m deep saturated zone in a MSW landfill established contaminant diffusion timescales from blocks of waste containing immobile fluid in the region of 50 to 100 days (Woodman et al., 2017) and, in seven tests at four different MSW landfills, between 1 and 300 days (Rees-White et al. 2021).

Less is known about contaminant transport in unsaturated conditions. As with saturated waste, preferential flow may leave areas of waste isolated from flushing. To overcome this, if the landfill design allows, it may be possible to flood the waste from the base up to ensure saturated conditions throughout. Clarke et al. (2016) describe a bioreactor trial in which two purpose-built waste cells were sequentially saturated and drained to promote degradation. Each 2 m deep test cell held ~1,000 t of shredded waste and were then subjected to ~40 flood and drain cycles over a period of nearly 300 days. Compared to an identical cell operated in unsaturated, down-flow conditions, the degradation of waste in the cells using the fill-and-draw method was accelerated. They did not assess the effects of repeated flushing on solute removal.

All flushing activities will require a significant volume of fluid to be added to the landfill, which will require treatment. This may have significant practical and financial implications to the type of flushing available. Lorton et al. (2008) describe the systematic flooding and draining of the Mont Cuet landfill, Guernsey in an attempt to supress a landfill fire. An undisclosed volume of groundwater was pumped into the ~5 Ha site to saturate the entire waste body to a depth of 10 m. At the end of the water injection period, leachate was pumped out and held in a lagoon for treatment. The project aim was to supress a landfill fire, not to remove contaminants. It does, however, demonstrate the feasibility and complications of flooding, and subsequently dewatering of an entire landfill with water, which is analogous to a single fill-and-draw cycle.

3 METHODOLOGY

3.1 Test cell

The fill-and-draw tracer test was carried out in the Landgraaf pilot test cell, The Netherlands (Oonk et al., 2013). The test cell was constructed in late 2000 as a raised structure above an existing landfill as part of a coordinated research programme to assess the feasibility of sustainable landfilling concepts, investigating methods to accelerate biodegradation and flush contaminants from the waste (Woelders et al. 2005, Oonk et al., 2013).

The as-built dimensions of the pilot cell were: Width 55 m, Length 80 m, Height 5 m at the edge and approximately 10 m in the centre (Figure 1). The cell was constructed with clay side walls, lined with a 2 mm-thick HDPE liner covering the base and sides, and installed with a basal leachate collection system. The cell was filled with ~25,000 tonnes of un-shredded domestic MSW and industrial wastes (which were pre-mixed prior to filling) at a density of ~1.08 t/m^3. Full waste characterisation can be found in Oonk et al. (2013). The top surface of the waste was covered with a permeable soil allowing infiltration, although pitched to encourage runoff.

Nine equally-spaced 125 mm-diameter slotted drainage pipes were installed across the base of the test cell. For the tracer test, only three of the drains were used, spaced approximately 20 m apart. The drainage pipes were buried in a 0.25 m-deep layer of coarse sand with a porosity of ~33% and a hydraulic conductivity of ~6×10^{-5} m/sec. The sand layer extended across the base of the landfill.

Six fully-penetrating, fully-screened 90 mm-diameter wells and six 25 mm-monitoring-piezometers at either deep or mid-level elevations and with a 0.5 m response zone were installed in the waste and were used for monitoring saturation levels and to take samples. Details of the wells and piezometers and more detailed site plans are given in Supplementary Information.

Between 2000 and 2005, a series of leachate recirculation trials were carried out in the pilot (Luning et al., 2006; Mathlener et al., 2006; Oonk et al., 2013) to fulfil the original research objectives of accelerating waste biodegradation. Leachate and water were irrigated and recirculated into the top of the cell via injection pipes installed in the top and mid-way up the sides (e.g. Vigneron et al., 2009).

3.2 Tracer test infrastructure

The water used in the test was injected using a centrifugal pump (Ebara, model JESX6), monitored with an electromagnetic flow meter (Endress & Hauser Promag 50) and controlled manually with a gate valve. Directly after the flow-control valve, a manifold split the flow into three 32 mm-pipes each leading to an injection point and into the basal drains

(M10, M30 and M50); flow into each injection point was balanced using manual control valves and monitored by datalogged mechanical flow meters.

A concentrated tracer solution was mixed and stored in a 1,010 L tank fitted with a motorised stirrer to ensure the tracer was evenly mixed. The concentrated tracer was injected into the pumped water at a rate of 1 ml/litre using a dosing pump (ProMinent gamma/ L Diaphragm Metering Pump) controlled by the output from the main flow meter. A schematic of the tracer injection set up is given in Supplementary Information (Figure SI 1).

The same infrastructure was used during the abstraction phase of the test. Midway through abstraction, during an enforced shut-down for 40 days, the centrifugal pump was replaced by a heavy-duty peristaltic pump (Watson Marlow Bredel SPX25) which was better suited to long-term abstraction at increasing suction pressures that resulted from dewatering the pilot.

3.3 Instrumentation and monitoring

Dataloggers (Campbell Scientific, CR1000 and Gemini, Tinytag) were used to measure flow rate and pressure in the delivery pipes. Logged pressure transducers were installed in four of the wells to monitor levels. The level in all wells and piezometers were also measured manually using a dip meter throughout the experiment.

Over 300 water samples were taken for laboratory analysis. Samples were obtained from wells, piezometers and the inlet and outlet lines (sample point S90 in Figure SI 1). Background samples of the indigenous leachate, collected before the tracer test was started, and samples of the concentrated tracer and the injected water were also analysed.

Samples were analysed for pH, ammonia, chloride, sodium and calcium using a Dionex DX 120 ion chromatograph. A filtered sub-sample was analysed for bromide and a range of ions using an ICP-MS (Thermo-Scientific X-series). Dissolved organic content (DOC) was measured using the TC/TIC difference method on an Analytik Jena 3100 analyser. EC was measured at constant temperature using a handheld conductivity meter (Mettler, Toledo FiveGo FG3).

Throughout the tracer test rainfall was measured at an on-site weather station.

3.4 Operation of tracer test

Prior to the start of the tracer test, leachate in the test cell was drained under gravity and by pumping from drain M50 (Figure 1). Settlement of the landfill and the elevation of the drain access point, meant that it was not possible to completely dewater the cell, and a 2.35 m saturated depth existed at the start of the test. Including the basal sand layer and assuming a total waste porosity of 40%, this equates to a starting leachate volume of ~2,900 m^3.

The injection phase lasted for 17 days. After a 3 day pause to alter pipework and instrumentation, abstraction commenced and was continuous for 26 days (Abstraction 1), at which point the test had to be interrupted for 40 days for site operational reasons. After the pause, abstraction was restarted and continued uninterrupted for a further 53 days (Abstraction 2) (Table 1).

3.4.1 Injection phase

A mass of 132 kg of KBr (88.8 kg Br) (extra-pure 98%, Acros Organics, Belgium) was mixed with water in the tracer supply tank to a total volume of 1,010 litres.

At the onset of the test, water dosed with concentrated tracer was pumped at a rate of 1.99 m^3/hour from the supply tank into the three drains. The corresponding tracer input concentration was ~89 mg/l Br. The average EC of the injected water was ~0.6 mS/cm. In total, 800 m^3 of tracer solution was injected over a period of 17 days, split equally between the drains. The relative proportions pumped into each drain were M10: 34%, M30: 33% and M50: 33% (Figure SI 2).

3.4.2 Abstraction phase

Abstraction commenced three days after injection had finished. It was initially planned to pump out at the same rate (1.99 m^3/hr) that tracer was injected. However, following 28 hours pumping at this rate it was determined that, in the longer term, this rate would not be sustainable as suction heads would develop in the system and at the pump. Pumping was switched off for 23 hours and then recommenced at a rate of 0.88 m^3/hr. As the pause was of similar length to the duration of the original pumping rate of 1.99 m^3/hr, a reasonable approximation is that pumping averaged 0.88 m^3/hr from the start.

Abstraction was continuous for 26 days during which time 556 m^3 had been pumped from the test cell. Following a 40-day shutdown, the flow rate was reduced to ~0.44 m^3/hr. Pumping continued for a further 52 days, during which another 542 m^3 was pumped from the site. In total, 1098 m^3 was pumped out over a period of 78 days of pumping. It was more difficult to keep the system balanced during abstraction compared with the injection phase. The relative proportions pumped from each drain at the end of the test were M10: 29%, M30: 31% and M50: 40% (Figure SI 3).

Injection and abstraction data are summarised in Table 1.

4 RESULTS & ANALYSIS

4.1 Background concentrations

Analysis of leachate samples taken from the six fully-penetrating wells prior to the start of the test are detailed in Table 2. The results show that leachate quality within the test cell was spatially variable.

4.2 Water levels

Figure 2 shows the average logged water level, normalised to mNAP (Amsterdam Ordnance Datum), measured in the wells throughout the test and average manual measurements from the piezometers. Full details of water levels in individual wells and piezometers are given in Figures SI 4 and SI 5. During tracer injection, water levels increased by an average of 2.2 m. Levels began to decline as soon as abstraction commenced and continued to decline at a linear rate until the 40-day break in abstraction. At the start of the break, water levels remained relatively static for a period of approximately two weeks, before increasing. Over the complete duration of the break, water levels in all the wells increased by ~0.70 m. The increase in head during the pause period is assumed to be recharge from surface water infiltration: 108 mm of precipitation was measured at the on-site weather station during this period, which equates to 528 m^3 of potential infiltration over the total surface area of the test cell, not accounting for run-off, ponding or evaporation. Precipitation in the early part of the pause period predominantly fell as snow, which persisted on the ground for several weeks, before melting at a time that correlates well with the start of the recorded increase in water levels.

Water levels decreased as soon as pumping was restarted and, after a further 53 days of pumping, had declined to within ~0.125 m of the level recorded at the start of the tracer test.

A similar piezometric response is seen in the piezometers although heads are somewhat lower, especially toward the end of abstraction. This may relate to the depth of the response zone and the distance from one or other of the three basal drains.

4.3 Water balance

Following the completion of the tracer test, the average water level in the test cell had returned to approximately its starting level, whereas 800 m^3 of water/tracer had been pumped in and 1,100 m^3 of leachate/tracer pumped out. The net addition of approximately 300 m^3 of water over the duration of the test is assumed to have occurred from rainwater infiltration. Although the top of the test cell was originally contoured to shed surface water, differential settlement resulted in large puddles of water collecting on the surface, particularly along the sides where the waste had settled below the edge of the bund. Direct evidence of infiltration was provided by the increase in water levels during the pause in abstraction, which correlated with meteorological data.

The increase in water level due to injecting 800 m^3 of water was calculated to result in the saturation of an additional 9,400 m^3 of waste. The change in volumetric water content (i.e. the fillable porosity of the waste) was calculated to be ~8.5%. The calculated drainable porosity of the waste is dependent on the amount of infiltration assumed to enter the cell but was between 8.5 and 11%.

4.4 Tracer responses in break-back curve

The concentration of the tracers Br and EC during the abstraction period, which we term the tracer Break Back Curve (BBC), are shown in Figure 4a and 4b. Concentrations of Br at the end of the BBC were approximately 18 mg/l, 11 % of the tracer average input pre-test background concentration. For EC, the tail of the BBC is 36% of the average tracer background conductivity. The tails of the BBC curves exhibit a certain amount of noise in the data, though the trend is the same for both tracers. This is attributed to variations in tracer concentration in the three individual drains (Figure SI 6), and differences in the volumetric mixing ratio from the three drains.

Figure 4c and d show the concentration of Cl and NH_4 measured in samples collected from the combined discharge. The concentration of Cl at the end of the BBC was ~815 mg/l, 81% of the average background. For NH_4, the end of test concentrations was ~280 mg/l, 37% of the background. Although a direct comparison of these values within the wide range of background concentrations is very qualitative, the data suggest that ammonia is behaving in a different way to chloride. The reason for this difference may in part be due to spatial variability in biologically induced degradation across the landfill cell.

4.5 Mass recovered

Results of mass balance calculated for the bromide tracer and the *in-situ* contaminants NH_4, Cl and other major ions Mg, Na, K and Ca are given in Table 3. Recovered solute mass was calculated by using the trapezium rule to integrate the break-back curve with respect to time,

$$\sum_n Q_n \, C_n \, \Delta t_n \qquad (1)$$

Where Q_n and C_n are the average flow rate and concentration measured in period *n* of length Δt_n. A comparison of injected versus recovered Br indicates that a recovery of 72% was achieved. However, some of the recovered tracer is from indigenous 'background' concentrations of bromide already in the site. The difficulty with establishing the proportion of tracer recovered is that there is some spatial variability in the background concentration, (Table 2), and it is not possible to differentiate in the laboratory analysis between the introduced bromide and the indigenous bromide. A correction to the bromide data was made by assuming an average background concentration of 10.1 mg/l in the indigenous leachate and by using the EC of the BBC to establish an approximate mixing ratio between the introduced tracer and the indigenous leachate. This correction indicates that nearer 63% of introduced Br was recovered.

At the end of the test, ~1100 m^3 of leachate had been removed. The estimated mass of NH_4 recovered was ~227 kg, and an estimated 579 kg of chloride was recovered.

5 NUMERICAL MODELLING

5.1 Flow geometry

A fill-and-draw cycle can be conceptualised as upwards vertical flow, which is then reversed. It is acknowledged, that the real geometry of the site was complex, but a simple conceptualisation of the flow pattern induced in the pilot cell is of water being fed uniformly through horizontal drains, spreading out through a drainage layer into the waste and moving a horizontal water table upwards. Justification of this simplified one-dimensional modelling approach is as follows: the inlet boundary condition for the test cell is of three horizontal drains positioned in a layer of sand beneath the waste, and water level measurements made during injection and abstraction demonstrate a relatively uniform water table across the pilot (Figures SI 4 & SI 5).

As leachate levels increased in the site, sampling from the discrete horizon piezometers showed the breakthrough of bromide (Figure SI 7) and the dilution of indigenous contaminants (Figure SI 8) in a manner consistent with the conceptual model of the introduced tracer pushing a front of indigenous leachate upwards.

5.2 Dual porosity modelling

The transport model used in this analysis simulates advection and dual-porosity exchange (i.e. neglecting mechanical dispersion). The dominant solute transport mechanisms are by advection in the mobile zone and diffusion into the immobile zone, characterised by a single spatially homogeneous block diffusion time.

The code used is called DP-PULSE (Barker et al., 2000; Fretwell et al., 2005), which simulates the movement of the water into/out of the landfill during filling and subsequent drainage in the presence of dual-porosity exchange. DP-PULSE has previously been used to analyse experimental data from laboratory (Woodman et al., 2007 and 2014) and field studies in MSW (Woodman, 2017, Rees-White, 2021).

DP-PULSE is a semi-analytical one-dimensional model, which represents the transport domain as a stack of cells. The solute (i.e. leachate and/or tracer) is passed from cell to cell in pulses, simulating advection. Between the pulses, solute will exchange from the mobile zone into an immobile zone by diffusion. A conceptual drawing of the model is given in Figure 3.

The model assumes that transport in the immobile zone is by diffusion alone and is described by Fick’s second law. Normally, simple geometries (slab, sphere, cylinder, etc.) are assumed for the immobile ‘blocks’ and this block shape can be regarded as a fitting factor

(Woodman et al, 2014); mixtures of block shapes can also be used (Barker, 1985). Since it is highly probable that there is a wide range of shapes and sizes of blocks, the simplest approach is to assume an average shape and group all the diverse blocks into a single equivalent which, for this analysis, was assumed to be slab-shaped (Barker, 1985). This is potentially realistic if the flow is being constrained along horizontal subparallel layers of low permeability plastic.

The waste is characterised by a porosity ratio, σ, the ratio of the immobile (θ_{im}) and mobile (θ_m) porosities (i.e., $\sigma = \theta_{im}/\theta_m$) (Barker et al., 2000), and t_{cf} the characteristic time for diffusion to/from the mobile zone. t_{cf} is defined as:

$$t_{cf} = \frac{a^2 D_a}{4D_e^2} \qquad (2)$$

Where a is the length dimension (m) of the width of the fracture and D_a is the apparent and D_e the effective diffusion coefficient, in units of area/time (Barker et al., 2000). Using these parameters, the block diffusion time, t_{cb}, can also be calculated:

$$t_{cb} = \sigma^2 t_{cf} \qquad (3)$$

t_{cb} characterises the period taken for contaminants to move diffusively out of the immobile zone under ideal conditions. For example, a block with slab geometry continually flushed by uncontaminated water will release ~93% of its solute mass in time t_{cb}. The analytical solution for this calculation is give in SI, Equation 6.

Analytical modelling has shown that due to the high permeability of the basal layer, flow in the waste is near vertical without significant variation with distance from the drain. For a spatially uniform rate of flow to the basal layer, the velocity of flow in this layer will vary linearly with distance from the drain and that in turn gives an exponentially distributed travel time between the drain and the base of the waste. For example, flow within the basal drainage layer is calculated to have a maximum travel time of ~3 days. Consequently, the basal layer transport will be equivalent to that of single mixing cell with the porous volume of the basal layer. Multiplying an estimated sand volume of 780 m^3 with a porosity of 33%, gives a porous volume of 257 m^3.

5.3 Model calibration

Piecewise-constant flow rates were used for the model and are given in Table 1. Concentrations and mass of introduced tracers were based on the data in Table 3. The solutes selected for modelling were the introduced tracers Br and EC, and two indigenous *in situ* contaminants, one conservative (Chloride, Cl) and one reactive (Ammonia, NH_4). Concentrations of these solutes obtained from samples of the BBC (in the discharge) were compared with model fits.

A major data input requirement of the model is the *in-situ* background concentration of the solutes. Table 2 already indicates considerable variation across the pilot, and this represents a significant uncertainty for modelling. A modelling assumption needed to be made that initial concentrations were spatially constant, and the average concentration measured in the six wells was used. This average will broadly encompass the variability of in situ concentrations that might be expected during the test.

The model was calibrated by minimising the sum of squares of errors (SSE) between observed and modelled parameters to derive the porosity ratio, σ, and $t_{cf,}$ from which the block diffusion time, t_{cb}, could be calculated using Equation 3. The initial background concentration, C_b, and the injected concentration, C_o, were fixed.

The best fit for the tracers Br and EC are shown in Figure 4a and 4b and for the indigenous contaminants NH_4 and Cl in Figure 4c and 4d.

In each case, the model fits the early (abstraction 1) data well but overestimates the tail (abstraction 2). The reason for the consistent over-estimation of the later data is likely to be due to recharge, which is not accounted for in the model. As discussed, during the course of the experiment an additional 300 m^3 of leachate was removed from the pilot for the same change in water level. This additional fluid is assumed to be from the infiltration of precipitation which mostly occurred during and after the pause in abstraction (Figure 2). Further evidence of recharge can be seen in the BBC data at the start and end of the 40-day pause. For the Br tracer, the concentration is lower at the end of the pause than at the start. If this was due to diffusion, by symmetry, it would be expected that the opposite would have occurred to EC, Cl and NH_4 during the pause. However, the indigenous contaminants also demonstrate a decrease in concentration during the pause, indicating dilution.

Based on the estimated recharge volume of ~300 m^3 from infiltrating rainfall and an estimated porous volume of ~4,000 m^3 at the end of Abstraction 1 a crude estimate of the potential dilution has been applied to the test data. Although this correction assumes unrealistically uniform dilution throughout the saturated waste, for the two tracers, the recharge-corrected data show a significantly better fit to the model. If the same recharge correction is applied to the indigenous contaminants the fit for Cl is not improved, but there is an improvement to the NH_4 fit. This may be due to the variability in pre-test concentrations measured across the pilot. The recharge-corrected data has not been used in the model fitting. However, it supports the hypothesis that there was dilution through recharge as a result of the infiltration of rainfall and snow melt.

Parameter estimates derived from the model fitting are given in Table 4. The porosity ratio, σ is relatively consistent between the two tracers, 15 for Br and 14 for EC. For indigenous contaminants, σ ranges between 10 and 6.7 for Cl and NH_4 respectively. A similar pattern is shown for t_{cf}, with shows consistency between the tracers and Cl, but a lower derived estimate for NH_4.

5.4 Simulation of flushing scenarios

The same model used to analyse the tracer test, combined with the derived parameters given in Table 4, has been applied to investigate the efficiency of flushing waste in the test cell using fill-and-draw. The analysis assumes that all the contaminant load exists in a soluble form within the matrix of the waste at the start of flushing.

Figure 5 compares the estimated removal of indigenous contaminants Cl and NH_4 at the Landgraaf pilot cell following repeated one month (i.e., 30 day) flushing cycles (15 days injection followed by 15 days abstraction). The analysis demonstrates that, carried out as described, the flushing efficiency of fill-and-draw is rather low and reduces with time. For both Cl and NH_4, it would take ~20 cycles (equivalent to ~2 years of continuous flushing) to reduce the load by 50%, and more than 60 cycles (~5 years) to reduce it by 75%.

The low flushing efficiency is partly a consequence of the long block diffusion times (i.e., t_{cb}), but also in part due to the way in which the fill-and-draw cycle operates. On injection, flushing liquid progressively saturates the waste from the base up and, during abstraction, the waste dewaters from the top down. During a fill-and-draw cycle, the lower waste will be saturated for the duration of the cycle, whereas the upper parts of the waste will only momentarily be saturated as the cycle reverses. This will disproportionately flush the lower waste compared to the upper waste, and is demonstrated in Figure 6. Here, the estimated mass remaining after a single flushing cycle of 15 days injection followed by 15 days abstraction is shown for the total depth of waste (using the maximum waste depth of 10m at the Landgraaf Pilot), where 0 m represents a water level at the base of the landfill. This shows that for both Cl and NH_4, no significant contaminant flushing occurs in the upper 4 to 6 m of the landfill. Increasing the number of flushing cycles improves the contaminant removal from the upper waste. However, even after 100 cycles (equivalent to ~8 years of continuous fill-and-draw cycles), more than 50% of the Cl and 30% of NH_4 will still remain in the upper parts of the landfill.

To increase the efficiency of flushing, particularly in the upper parts of the waste, a period of ‘shut-in’ may be necessary. In this scenario, at the end of the injection period the waste is left fully-saturated for a given duration to allow time for the diffusion of contaminants to occur, before the waste is then drained. The influence of including shut-in on contaminant removal is also examined in Figure 5 which compares the flushing regime of 15 days injection/15 days abstraction with an equivalent cycle that also includes a 15-day period of shut-in (total 45 days). Including a period of shut-in increases the amount of contaminants removed, although in the example given, it has more influence on the Cl data which has a significantly longer block diffusion time.

Figure 5 also shows the result of reducing the injection and abstraction time, but maintaining the same 15-day shut-in period (7.5-day injection and 7.5-day abstraction, with a 15-day shut-in, total 30-days). The analysis shows, that a 30-day cycle that includes a shut-in is more efficient at removing contaminant mass than a 30-day injection/abstraction cycle only, even though the overall duration of the cycle is the same. This is because the shorter abstraction phase allows less time for contaminants that have diffused into the mobile water at the top of the waste column, and which have a higher concentration than in the mobile zone at the base of the column, to diffuse back out. An instantaneous draining of the mobile water, following a period of shut-in, would be the most efficient method, although this would not be achievable in reality.

To find the optimum duration for a shut-in period, the flushing scenarios examined in Figures 5 and 6 are compared with an increasing period of shut-in over 10 flushing cycles, Figure 7. For Cl, there is no significant improvement to the mass recovered for shut-in periods greater than ~60 days. For NH_4, there is no further improvement after ~7 days. The optimum shut-in period will be a function of the block diffusion time, with shorter values of t_{cb} requiring shorter shut-in periods.

A further consideration when designing a flushing regime, will be the ratio of immobile to mobile porosities, represented by σ. At Landgraaf, σ ranged from 7 to 15. This is within the range measured in other full-scale tracer tests carried out in waste at other MSW landfills where σ was estimated to range between 4 to 20 (Rees-White et al., 2021). The estimated mass removal for different values of t_{cb} and σ for twelve fill-and-draw cycles of 15 days injection, 15 days shut-in, 15-day abstraction is compared in Figure 8. This demonstrates that the lower the value of σ, the greater the contaminant load that will be removed over the same duration of flushing.

In general, the mass removed increases with the number of cycles, the length of each cycle and with σ; but decreases with t_{cb}. As the cycle length increases the percentage of mass remaining after N cycles tends to an upper limit given by:

$$P_N(\sigma) = 100\left(1 - \frac{1}{\sigma}\right)^N \quad (4)$$

This equation could be used, for example, to provide an optimistic estimate of the minimum number of flushing cycles required.

6 DISCUSSION

The highly controlled tracer test allowed the recovery of contaminant transport properties of the waste based on the analysis of both the introduced tracer (Br), and the response of indigenous contaminants (EC, Cl and NH_4) in the waste and leachate. The advantage in tracing the injected water is that the input concentration (mass) is well known. This contrasts with the noted uncertainty in indigenous contaminant concentrations. Although there was some variation in the background concentration of bromide, it was low relative to the mass injected and a uniform distribution of tracer was also measured in samples collected from piezometers installed in discrete waste horizons.

The best fit for Br derived a block diffusion time of 328 days, more than twice the duration of the experiment (137 days). A similar t_{cb} was calculated from analysing the EC data (278 days) and Cl data (229 days). Since EC is a parameter that is easy and cheap to monitor and lends itself to being datalogged, it is advisable to monitor EC when clean water is being used as a tracer in landfills. Previous estimations of the block diffusion time in well-constrained column experiments have been shown to be between 5 and 100 days (120 and 2,400 hours, Woodman, 2007) and, in field studies, between 5 and 275 days (120 and 6,600 hours, Woodman et al. 2017, Rees-White et al., 2021).

The calculated t_{cb} for NH_4 (28 days) was an order of magnitude lower than for the other species. If the system was behaving ideally, this block diffusion time would be expected to be of the same order of magnitude. The difference may, in part, be due to the spatial distribution of background concentration which was very poorly constrained. This was particularly so for NH_4 which had a greater range in measured concentrations (StDEV 49% of average) compared to Br and Cl (StDEV 33% and 32% of average).

Using the derived value for t_{cb}, the length-scale of the immobile zone can be calculated based on the definition of t_{cb}. For a slab, the dimension, *b*, is the half width of the slab and it is related to t_{cb} as

$$t_{cb} = \frac{b^2}{D_a} \qquad (5)$$

where D_a is the apparent diffusion coefficient (Barker et al., 2000).

Using a free-water diffusion coefficient for bromide of between 1.85×10^{-9} m^2/s (Leistra & Boesten, 2010) and 2.1×10^{-9} m^2/s (Cussler, 2009) as an estimate of the apparent diffusion coefficient of bromide in leachate, gives a broad estimate of the block (half-width) dimension *b* to be calculated. Applying Equation 5 to the Br data, gives a range of *b* between 0.21 and 0.24 m. For chloride, using an average diffusion coefficient of 1.53×10^{-9} m^2/s (Tang and Sandall, 1985), *b* is ~0.17 m. This is within the same range as Br, giving further confidence in the model fitting. Although relatively small on the landfill scale, blocks of this size (2*b* = 0.34 to 0.48 m) will result in sections of the waste which are isolated from the surrounding flow.

Predictive application of the model, using parameters obtained from calibration against the test data, suggests that the most efficient fill-and-draw strategy in terms of the least number of cycles, and hence water used, would be for the fill and then draw to both occur instantaneously, combined with an optimum shut-in period, as examined in Figure 7. Instantaneous fill or draw is physically impossible. In reality, the duration of the fill cycle will likely be constrained by the availability of water required to saturate the site, the infrastructure used to inject it and the rate at which the water is taken up by the waste. At Landgraaf, we were restricted to a maximum injection rate of ~2 m^3/hour which was a limitation of the supply of water. For the draw cycle, the draining rate may be limited by the available treatment or storage options for the leachate being recovered. In the test discussed, we were not able to maintain the same abstraction rate as during injection due to limitations in pumping infrastructure.

There are advantages of fill-and-draw over other forms of flushing (such as infiltration or well-to-well flushing (Woodman, 2017, Rees-White, 2021). In particular, the upward migration of a clean-water saturation front into unsaturated waste will largely overcome problems of preferential flow.

Any flushing technique will require fluids with low initial solute concentrations to inject into the landfill. In the case of fill-and-draw, the volume of water required to saturate a landfill, and volume of leachate generated on dewatering, would be significant and may make the technique impracticable or uneconomical at all but the smallest of sites. Furthermore, unless the entire waste body is saturated, waste in the upper parts of the landfill, above the fill level, will remain untreated. To overcome this, a combination of flushing techniques, utilising both saturated and unsaturated methods, may need to be used, for example, alternating between fill-and-draw and flushing from above through irrigation.

The tracer test demonstrates the practical feasibility of using an engineered basal drainage system to carry out fill-and-draw flushing in suitable landfills. The characteristics of a suitable landfill are likely to include a secure hydrogeological situation, high standards of engineered containment, and good vertical waste permeability. The latter requirement will mostly limit fill-and-draw flushing to shallow landfills as there is a strong inverse correlation between waste hydraulic conductivity and effective stress (e.g. Powrie and Beaven, 1999).

7 SUMMARY AND CONCLUSIONS

A single-cycle fill-and-draw tracer and flushing test was demonstrated as being operationally viable within a 10-metre-deep landfill cell. A dual-porosity model was calibrated against the field data and then used to investigate alternative operational approaches to fill-and-draw flushing.

The introduction of 800 m^3 of water containing the tracer bromide into the basal drainage system of a landfill test cell over a period of 17 days resulted in a uniform increase in leachate head across the site and the saturation of an additional 9,400 m^3 of waste. Abstraction took place in two phases of 24 and 52 days, separated by a 40-day period during which there was no abstraction. In total, 1100 m^3 of tracer/leachate was pumped from the base of the landfill. It is estimated that over the period of the experiment, an additional 300 m^3 of water was added to the landfill through the infiltration of precipitation.

The introduced tracers (Br), and indigenous contaminants (Cl, NH_4-N) diluted by the water used as a carrier for the Br, were suitable as tracers. The water was also used as a tracer, measuring leachate dilution using Electrical Conductivity. The advantage of using an introduced tracer like Br, is that the input concentration (mass) is well known. Tracer recovery was between 63 and 72% for bromide. This contrasts with the uncertainty in indigenous background contaminant concentrations where a mass balance is not possible. Nevertheless, this study demonstrated the benefit of using diluted indigenous contaminants as tracers, with EC being recognised as a tracer that is easy and cheap in monitor.

The recovered contaminant transport values of the landfill were consistent with other studies. Block diffusion times in the order of one year were estimated, although NH_4 behaved differently to the other solutes examined.

Based on the recovered contaminant transport parameter values, modelling indicates that many cycles of fill and draw would be required to significantly reduce the contaminant load of landfills by flushing. In particular, it is waste nearer to the top of the fill cycle that is difficult to flush. Limiting factors to the contaminant removal efficiency include long contaminant diffusion times from immobile blocks of waste. Including a shut-in time between the fill-and-draw cycles improves mass removal, although the influence decreases as the shut-in time increases.

The results of this trial demonstrate the ‘fill-and-draw’ concept using the basal leachate drainage system. Fill-and-draw requires a significant volume of water to saturate the waste mass, and this water will require storage and/or treatment on abstraction. This may make the technique impractical or uneconomical for many operators. Since fill-and-draw targets flushing in the lower part of a landfill and more conventional infiltration or recirculation schemes impact the top of the waste, there may be scope to benefit from each in combined schemes.

The models that have been developed will form the basis of future design tools. Further work is required to assess to efficiency of this technique in comparison to and in combination with other methods of landfill flushing.

ACKNOWLEDGEMENTS

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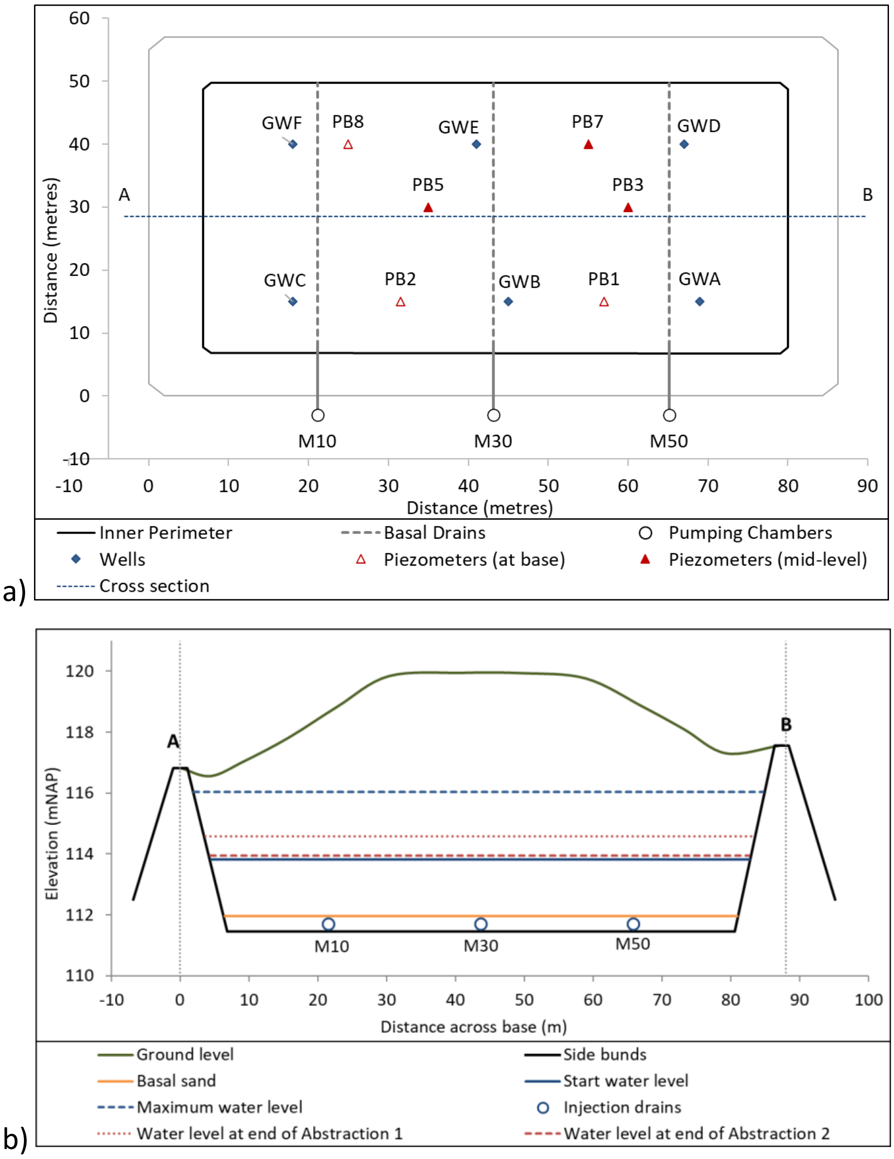

Figure 1. Pilot test cell: (a) plan of pilot test cell showing location of pumping chambers and monitoring points used in the tracer test; (b) cross-section through test cell showing the average start, end and maximum water level, ground level and location of the drains used for injection (mNAP = Amsterdam Ordnance Datum)

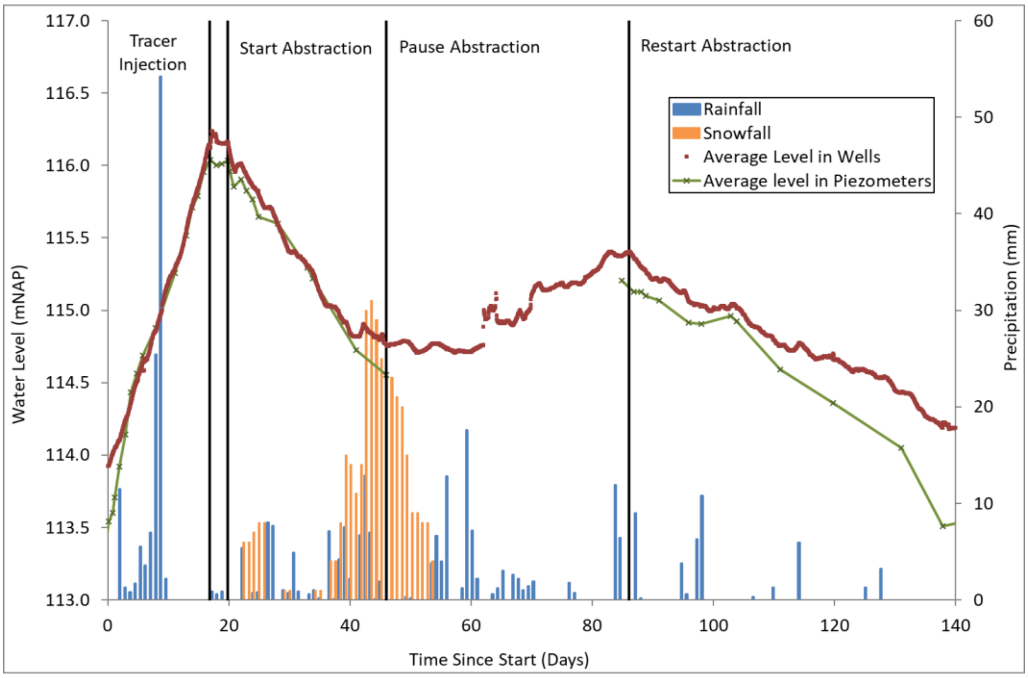

Figure 2. Average water level (normalised to mNAP) in wells and piezometers, with total daily rain and snowfall in mm. The average base of the landfill basal drainage layer was ~111.5 mNAP.

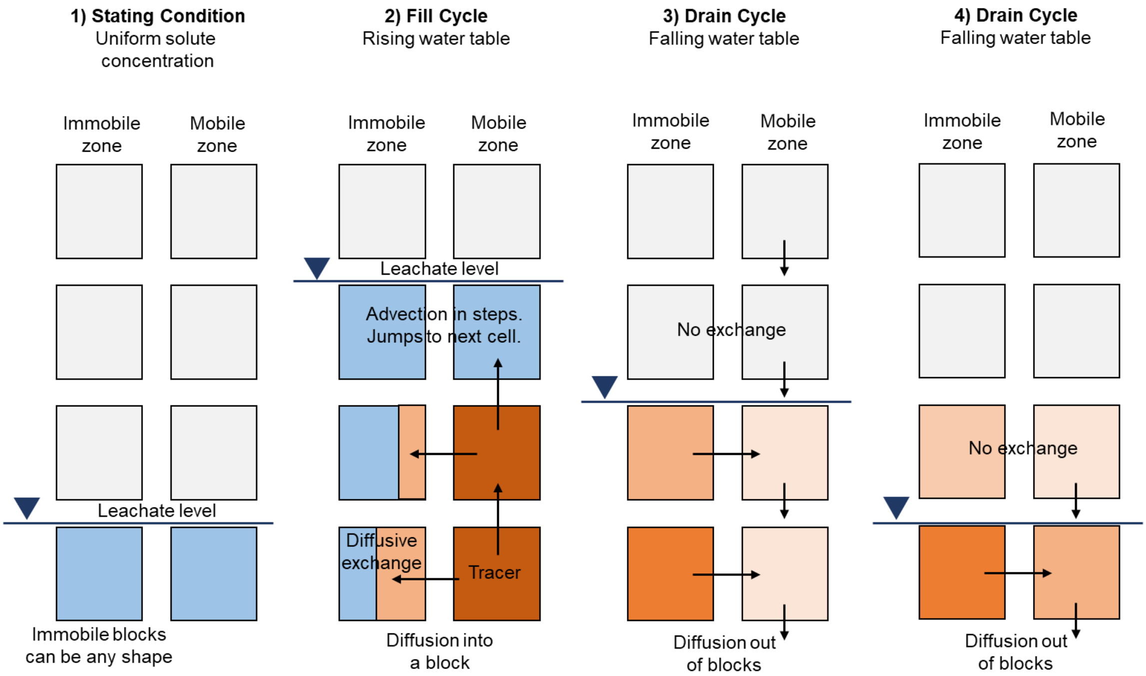

Figure 3. The DP-PULSE concept applied to a rising and falling water table in a fill-and-draw test

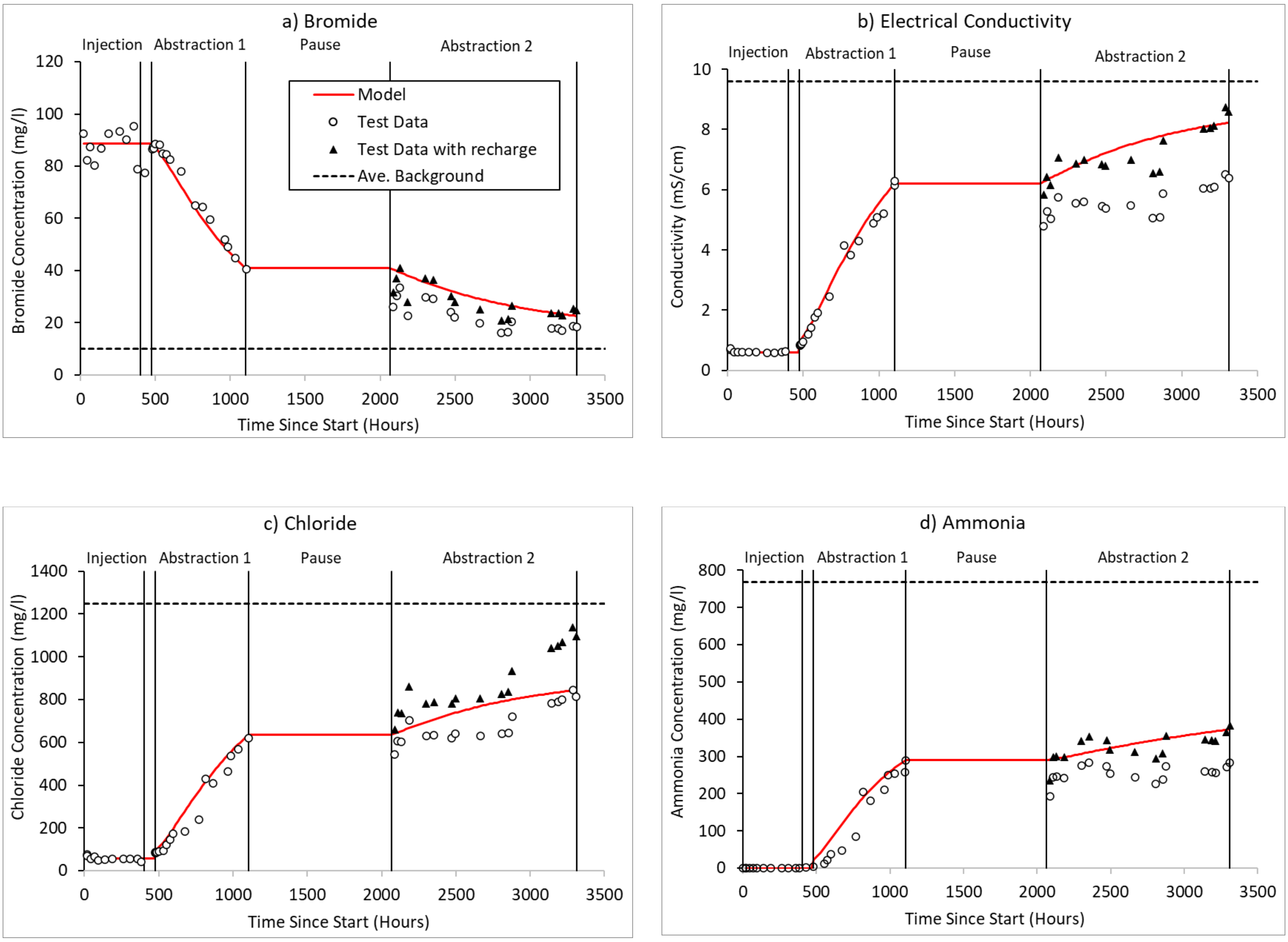

Figure 4. BBCs and model best fits to abstraction data for a) bromide, b) electrical conductivity, c) chloride, d) ammonia. Plots show injection and abstractions periods and average background concentration. Time is from the start of tracer injection. Estimated concentrations corrected for recharge are shown but were not used in model fitting.

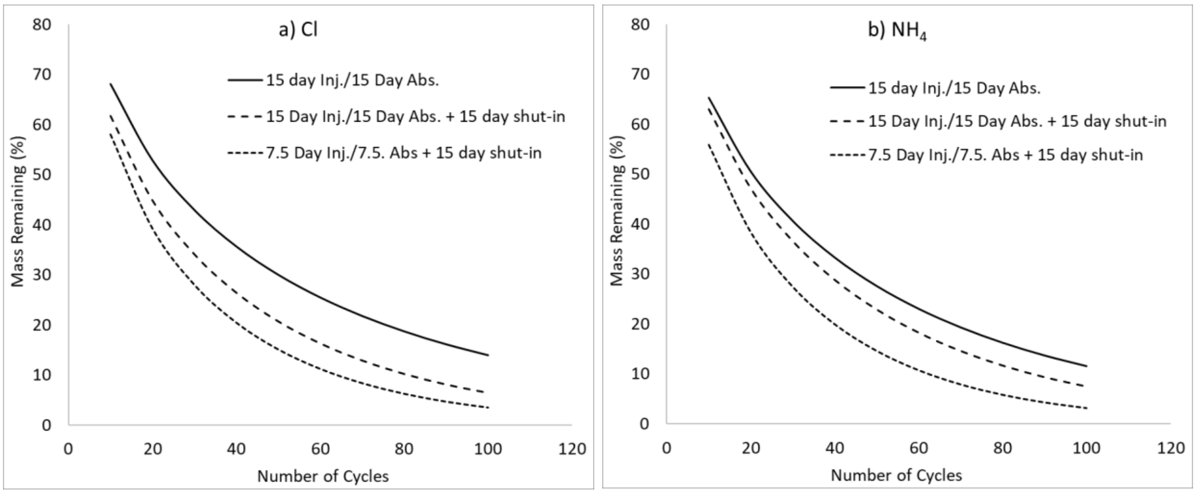

Figure 5. Contaminant load remaining after a given number of fill-and-draw cycles and duration for a) Cl and b) NH_4, and the influence of including a period of shut-in in the flushing cycle

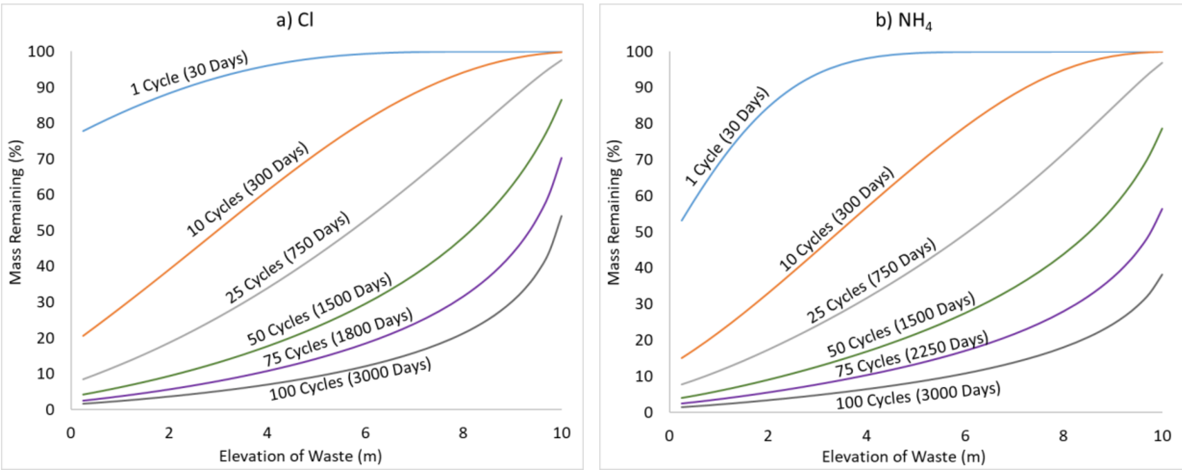

Figure 6. Contaminant load remaining for a given elevation of waste in a 10 m deep landfill for Cl and NH_4. Data shown for an increasing number of flushing cycles from 1 to 100

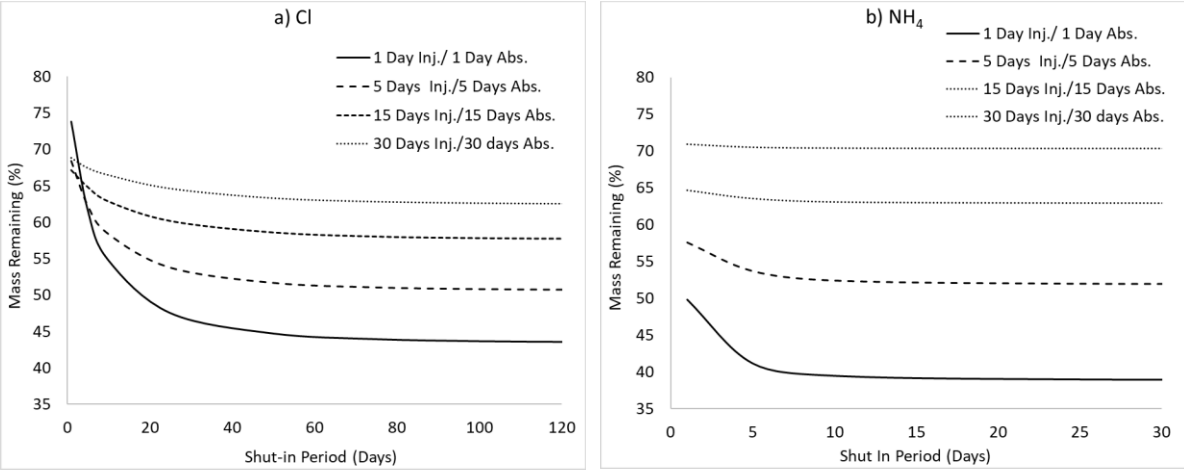

Figure 7. Comparison of increasing shut-in period on a) Cl and b) NH_4 mass removal for fill-and-draw regimes for 10 cycles of either 1-, 5-, 15- and 30-days injection followed by the same period of abstraction

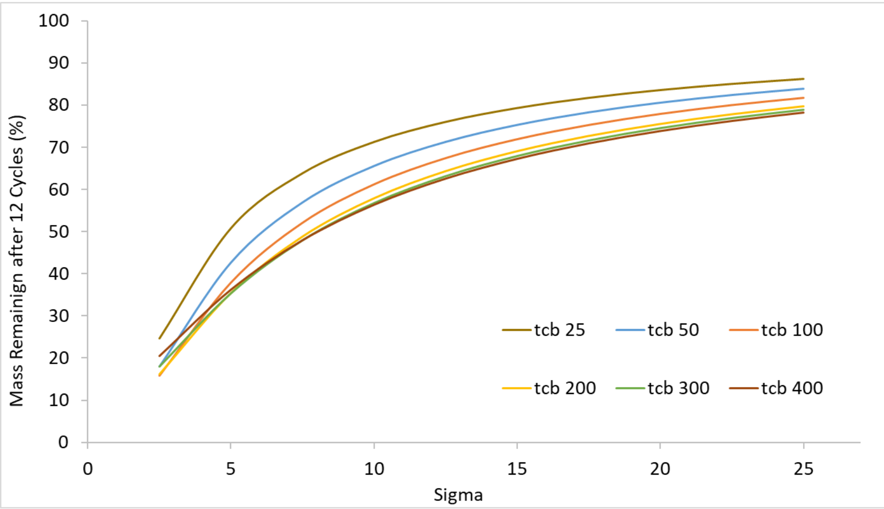

Figure 8. Comparison of flushing potential at different values of t_{cb} and σ (15 days injection, 15-day shut-in, 15-day abstraction over 12 cycles).

Table 1. Test summary data giving flow rates, duration and volumes pumped for each phase of the test

	Duration	Average Flow Rate	Volume Pumped	Proportion of flow in		
				M10	M30	M50
	days	m^3/hour	m^3	%	%	%
Injection	16.8	1.99	800	33.9	33.3	32.8
Pause	3.1	0.0	0	-	-	-
Abstraction 1	26.2	0.88	556	34.3	30.3	35.4
Pause	40.0	0.0	0	-	-	-
Abstraction 2	51.9	0.44	542	22.0	31.4	46.6
Abstraction 1 + 2 Combined	78.1	0.58	1098	29.0	30.8	40.3

Table 2. Background leachate concentrations measured in the six fully-penetrating wells

Parameter	Average	Min	Max	StDEV	StDEV as % of average
pH	7.74	7.49	7.92	0.1	2%
EC (mS/cm)	9.67	6.83	15.4	2.7	28%
Bromide (mg/l)	10.1	6.8	15.9	3.3	33%
Chloride (mg/l)	1007	643	1657	323	32%
Ammonia (mg/l)	768	412	1562	377	49%
Li (mg/l)	0.58	0.2	1.0	0.26	46%
Potassium (mg/l)	408	272	826	191	47%
Magnesium (mg/l)	66.1	47.6	84.1	12.9	20%
Sodium (mg/l)	1159	713	2181	477	41%
Calcium (mg/l)	57.7	31.8	80.2	20.1	35%
DOC (mg/l)	531	328	895	202	38%

Table 3. Summary of injected and abstracted tracer and ion masses

	Averaged injection concentration mg/l	Mass injected kg	Mass pumped out kg	Total mass recovered kg (% recovered)
Br	87.0	69.6	50.3	50 (72%)
Br (EC corrected)	87.0	69.6	50.3	44 (63%)
NH_4	0.09	0.07	227	227
Cl	64.3	51.4	630	579
DOC	13.8	11.0	201	190
Mg	9.3	7.4	44.1	37
Na	21.8	17.4	617	600
K	41.8	33.4	274	240
Ca	54.4	43.5	870	827

Table 4. Model input parameters C_o (the injected concentration) and C_b (the initial background concentration), derived values of t_{cf} and σ, and calculated t_{cb}

	C_0 mg/l or mS/cm	**C_b mg/l or mS/cm**	**t_{cf} Hours**	**σ**	**t_{cb} Hours**	**t_{cb} Days**
Br	92.0	10.1	35	15	7875	328
EC	0.6	9.6	34	14	6664	278
Cl	57	1248	55	10	5500	229
NH_4	0.01	768	15	6.7	675	28