

Quantifying the Solute Transport Water in a MSW Fill in Varied Saturated Conditions with Different Tracers

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Abstract: The fractional volume of the water involved in the transport of solute in a municipal solid waste (MSW) fill was estimated from the flux concentration of multiple tracers - Lithium, Sodium Chloride, Bromide and Brilliant Blue. The tests were undertaken in saturated and oversaturated conditions of the waste and the solute transport was driven entirely by gravitational forces. Mechanical mixing and transport of the solutes in preferential pathways was confirmed with the observation of relatively early modal concentration of solutes in the waste fill. The fitting of the lifetime probability density function (pdf) of the travel time of solutes to the normalised flow concentration suggests that the test result is not significantly influenced by experimental scale. The percentage of the water involved in the transport of solutes through the waste fill is estimated to be approximately 98%, of which approximately 80% appears to be involved in fast "advective" flow. Unlike soil grains, the waste constituent elements appear to act as dynamic stores and permeable media, which significantly improve solute and water mobility within the configuration of the emplaced waste. [Journal of American Science 2010;6(9):419-430]. (ISSN: 1545-1003).

Keywords: breakthrough curve, modelling, municipal solid waste, preferential flow, solute transport, tracer

1. Introduction

Landfills have been and will continue to be the primary method of waste disposal in the world. Approximately 70% of the global household waste produced is disposed of to landfills (OECD, 2001). As a consequence of increasing public awareness on the environment, the arbitrary dumping of waste materials, which is still prevalent in the developing countries, has virtually ceased in the Western countries in recent years. In addition, the waste regulators in these countries have continued to enact laws that aim to further reduce the volume of biodegradable streams sent to landfills. In the United Kingdom (UK) for instance, the "Landfill Directive" has positively imparted the rate of recycling in all the counties to a degree that previously seemed unattainable. In the past five years, the amount of waste sent for recycling (33.8 million tonnes) has nearly doubled (DEFRA, 2008). Nevertheless, the environmental issues with old unregulated landfills and existing operating ones that still contain biodegradable constituents will remain if efforts are not made to control the rate of temporal biodegradation within these fills. As suitable lands for physical development becomes dearer, it becomes imperative to gear efforts to accelerate the stabilisation processes in waste landfills to enable sustainable post-closure use. "Leachate", probably the most common term associated with landfills by the public is produced from the leaching of contaminant solutes in the waste constituents by the flow of water

in the waste fill. The moisture stored and its ensuing volumetric flow within a waste fill is probably the most important factor that influences the general long-term stability of a landfill, and therefore suitability for physical development (Oni, 2000; El-Fadel *et al.*, 1997; Rosqvist and Destouni, 2000; Li and Zeiss, 2001; Gawande *et al.*, 2003; Beaven *et al.*, 2005; Rosqvist *et al.*, 2005). The water in the landfill enhances biochemical processes that influence gaseous emissions and quality of leachate in the waste fill. In essence, appropriate management of the water flow within the waste can be used to level out the energy and the gradients of contaminant solute concentration between the landfill and the natural environment in a controlled way within an acceptable time limit (Bendz *et al.*, 1998).

Accurate modelling of the leachate produced from waste fills requires realistic simulation of the solute transport within the bulk motion of the water within the waste fill. In the majority of modelling of leachate, the models were usually an adaptation of models initially created for soils and modellers found it convenient to consider the waste fill in the macroscopic scale, as a homogenous medium (Bleiker *et al.*, 1995; Oni and Richards, 2004a). It is commonly accepted that waste is a complex material and accurate prediction of leachate quantity and quality is a highly complex task due to inadequate understanding of the spatial and temporal variation of the moisture flow in the waste fill (El-Fadel *et al.*, 1997; Beaven, 2000; Oni and Richards, 2004b). Identifying the flow pattern

and volume of solute transport water in a waste fill will undoubtedly improve the efficiency of a waste recirculation system via accelerated biodegradation. Waste by composition, source and method of emplacement is heterogeneous, consequently, there is heterogeneity in and within the daily lifts and cells of the emplaced waste. As the internal configuration of a medium will influence the flow pattern, it is therefore not surprising that the flow through a waste fill has been found to be non-uniform (Bendz *et al.*, 1998; Rosqvist and Bendz, 1999; Beaven and Hudson, 2003; Woodman *et al.*, 2005; Rosqvist *et al.*, 2005). Recent studies involve the numerical modelling of the mass flow of leachate, in an effort to understand the leachate solute flux in waste fills (Oni, 2010).

The modelling of solute flow through soils is prevalent in the literature (Jury, 1982; Jury and Sposito, 1985; Sposito *et al.*, 1986; White *et al.*, 1986; Butters *et al.*, 1989; Brusseau and Rao, 1990; Coppola *et al.*, 2009), however, the modelling of solute flow through waste is relatively modest. In both cases, the concept of non-ideal flow called "preferential flow" occurring as a result of local-scale heterogeneities is a major factor considered in the model formulation. The theoretical background of preferential flow, established more than 50 years ago, and initially applicable to flow in fissured rocks, has since been found to be applicable to porous medium comprising cracks and macropores (Barenblatt *et al.*, 1960; Jury, 1982; Coppola *et al.*, 2009). Preferential flow comprises of all occurrences where water and solutes move along certain pathways while bypassing other fraction of the porous medium (Coppola *et al.*, 2009). In soil science, the main concern of preferential flow is the tendency to enhance leaching of pollutants into the deep layers of the soil formation, where the natural biological degradation is weak by bypassing the bacteria-rich upper zone where natural attenuation of the infiltrating contaminants occur. In waste, however, the main concern about preferential flow is the tendency to cause partial leaching of the bulk waste fill, and by doing so, give false indication of the pollution potential of the waste fill (Rosqvist *et al.*, 2005).

Advances in the research and formulation of pragmatic models for preferential flow in soils have suggested the notions of mobile and immobile water, dual porosity, dual permeability and the consequential non-equilibrium flow in porous medium (Gaudet *et al.*, 1977; White *et al.*, 1986; Gerke and van Genuchten, 1993; Šim nek *et al.*, 2003). In the studies on waste, however, the modelling of solute transport has been undertaken with the basic ideology of a fast advective transport co-existing with a much slower transport of solute in the areas that are not on the preferential path (Rosqvist, 1999; Rosqvist and Bendz,

1999; Rosqvist *et al.*, 2005). It is generally believed that solute is carried through a mobile water fraction in fissures (or macropores) that lie adjacent and separates waste elements that contain the immobile water fraction and that diffusion occurs between the waste elements and the fissures (Beaven and Hudson, 2003). Transfer function analysis has been used to estimate the expected solute travel time and modal travel time and the corresponding solute transport fraction from the waste flow data obtained from test cells and experimental landfill (Rosqvist and Bendz, 1999; Rosqvist and Destouni, 2000). In addition, Lagrangian modelling, which ignore molecular diffusion has also been applied to waste, in an effort to accurately fit model results with field data.

Tracer tests that are traditionally used for soil studies are now often used in investigations involving flow patterns in waste. In order to identify the true path of flowing water, the chosen tracer must be chemically conservative, with the only transport processes influencing a conservative tracer being advection, dispersion or /and diffusion. It is therefore imperative to know the background chemical composition of the medium being examined to be able to choose the appropriate conservative tracer. Various fluorescent dyes and halides such as chlorides, bromides and iodide are often used. Lithium is the most popular tracer of landfill leachate owing to its usually undetectable quantities and minimal interaction with the waste (Öman and Rosqvist, 1999; Pocachard, 2005).

In this study, the relationship between , the lifetime probability density function (pdf) of the solute transport, which have been previously used on soil (Jury, 1982) and the normalised mass flux (White *et al.*, 1986) have been used in estimating the bulk average solute travel time and consequently the volumetric fraction of the solute transport water in a waste fill. Unlike previous studies (Rosqvist, 1999; Rosqvist and Bendz, 1999), direct analytical method of estimating the solute travel time have also been employed to improve the quantification of the solute water flux. Furthermore, the transfer function model (TFM) was applied to the different data obtained using various tracers for both saturated and oversaturated conditions of the MSW fill. This approach is also simple and pragmatic than previous modelling involving applied mathematical manipulations where multi-modal modelling is often assumed even though it is not evident in the characteristic trend of the measured concentration of effluents from the waste fills. Throughout this paper, the water transporting the solutes gravitationally in the waste fill is generally assumed to be vertical.

2. Methodology

2.1 Experimental data

2.1.1 The test cell

The test cell had been previously used as a Consolidating Anaerobic Reactor (CAR) for MSW materials (Ivanova, 2003). Suitable adjustments such as the dismantling of the anaerobic structure were however undertaken to enable the accomplishment of the purpose of the current study. The cell unit (Figure 1) consists of a cast acrylic cell, 0.480m inner diameter, 0.90m high, and 12mm wall thickness mounted in a sturdy framework. Inside the cell was a circular galvanised iron mesh, placed on the surface of the waste fill to prevent vertical movement of lightweight particles during the flow of water in the cell. There are three ports located at the side of the cell at approximately 100mm, 250mm, and 400mm above the base respectively. Vertical tubes attached to these ports served as manometers. Sixteen radial holes, 12mm diameter, were drilled uniformly on the top platen, serving as inflow channels of water and tracer into the surface of the waste. It also enables free vertical flow under gravity through the waste. Furthermore, eight similar holes were drilled uniformly on the base plate to serve as outflow channels of water from the sample.

Water was pumped into the cell from a 50-litre water container with two peristaltic pumps. The pumps had been calibrated to enable the required inflow rate (1l/hr) to be easily attained. The water supply distribution system comprised eight 3mm diameter silicon tubes connected to the inner ring of holes on the top platen and additional 8 tubes connected to the holes on two outer rings. In this way, the inflow was evenly distributed. The total length of these tubes from the pumps was approximately 1.7m. Water flow from each outlet port was conveyed through a 3mm diameter silicon tube, 1m long, connected to a lower end of an inverted Y piece-connector, which was fastened to an elevated horizontal bar. The bar was firmly fixed onto two vertical metal stands. The top end of the inverted Y connectors was open with a free water surface, 3.5 cm below the surface of the waste fill. The openness of the top of the inverted Y connector enabled atmospheric pressure to drive the flow down the other lower end of the connector to the auto-sampler. An improvised stilling well was improvised and fixed between the automatic sampler and the Y connector to serve as an atmospheric breaker owing to occasional bubbles that may cause water spills in the flow system, as experienced during the pre-test trials. The outlet tubes from the cell were formed into a common channel as the inflow into the auto-sampler is restricted to one flow channel. Load cells were fixed to the entire cell to measure changes in the mass and volume of the water in the waste fill with time. For

data logging, probes were located on the surface of the waste fill, in the pond, and also between the outlet port and the automatic sampler. To maintain good mixing of the water in the surface pond, an electric stirrer was fastened to the frame on the top of the pond.

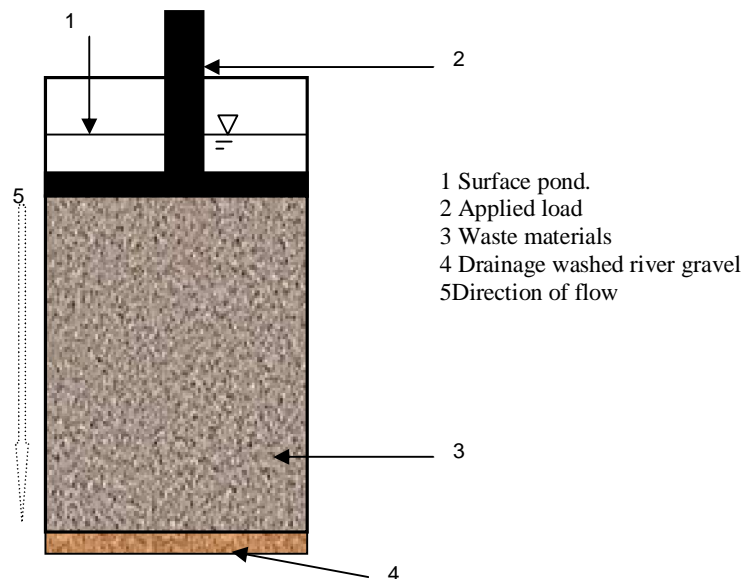


Figure 1: Experimental set-up for oversaturated flow condition.

2.1.2 Test material

The waste materials used in these tests consist of a 15-year-old MSW obtained from a landfill site at Rainham, Essex, UK. Large samples were obtained from the site and classified as shown in Table 1 (Ivanova, 2003; Beaven et al, 2005). Prior to the infill of the waste in the test cell, the sample was shredded to a maximum particle size of 40-50cm and then recomposed to the representative fraction of constituents determined in the initial physical classification. The waste overlays approximately 0.1m layer of 10mm washed river gravel in the cell. The cell was filled with 56.2kg (dry mass) of the waste and 90 litres of synthetic leachate, which contains 10% anaerobically digested sewage sludge. The waste was degraded at an applied load of 47kPa under anaerobic conditions for 338 days during which the factors affecting rate and magnitude of secondary settlement in the waste were studied. Under these prevailing conditions, the height of the waste fill decreased from 0.623m to 0.572m which is equivalent to an increase in dry density from 498kg/m³ to 572kg/m³.

2.1.3 Test procedure

The field capacity of the waste fill was determined from the volume of water absorbed in the waste body prior to gravitational flow. Subsequently,

the waste was washed out thoroughly of inherent chemical compounds from previous tests and left to drain completely. Chemical analysis of the water bubbles noticed during preliminary water flow indicated merely entrapped air. The top platen was lowered down on the waste fill to apply a load of 150kPa until there was no significant rebound. Consequently, the height of the waste fill in the cell decreased to 0.43 m. The drainable porosity, which is a measure of the effective water that can be drained from a saturated waste body under gravity, were determined by sequential saturation of the waste in both vertical directions. The water levels in the manometer tubes, representing the hydraulic level, at each water increment were measured as soon as the levels stabilised. The drainable porosity was then calculated from the gradient of the linear plot of the data of hydraulic head against the volume of the leachate drained; as opposed to the traditional method

of dividing the volume of water drained by the total volume of the waste fill. In most cases, the traditional method is susceptible to the errors caused by incomplete water drainage from the waste body. Water was re-circulated in the cell at a flow rate of 1l/h until a steady state flow condition was established. The ensuing volume of surface pond that enabled the gravitational flow rate of 1l/h was marked on the cell wall to enable the monitoring of steady state conditions during the tests. The volume was also measured to enable the volume of tracer needed for the tracer tests to be calculated. Furthermore, the hydraulic gradient and flow rate were measured to enable the saturated hydraulic conductivity to be calculated using the Darcy's law. In order to reduce the entrapped air in the waste fill, the recirculation of the water continued at the steady state for 3 days prior to the start of tracer tests

Table 1: Composition of the waste fill

Waste component	Paper	Plastic	Textile	Wood	Glass	Metal	Other
Percent (dry mass)	9.6	16.67	3.86	3.78	3.65	1.78	60.66

The tracers used in the solute transport tests comprises four common tracers; namely Lithium (30mg/l), Bromide (345mg/l), Coomassie Brilliant Blue (0.1g/l) and Sodium Chloride (5g/l). They were mixed thoroughly with water in a 50l-container until total dilution appeared to have been obtained. The concentrations of the tracers were measured from six samples obtained from the bulk solution to ensure the design concentrations were attained in the input supply. Prior to the start of the tracer test, four samples of the outflow from the cell were collected in 25ml bottles to establish the background concentration for each of the tracer in the waste fill. The concentration of Bromide was indirectly measured with an ion selective electrode while the concentration of Lithium was determined using an Atomic Absorption Spectrophotometer. The EC of Sodium Chloride was measured using a data-logger and hand probe.

As the test commenced, the peristaltic pump tubes inserted in the water supply container were interchanged into the tracer-solution supply container for continuous sprinkling of the tracer at steady state. The tracer supply was in "top hat" pulses of 1l/h on the surface pond overlying the waste thereby creating an oversaturation condition in the waste fill in the cell. The water flow from the waste fill was collected in 1-litre glass containers every hour by an auto-sampler. Tracer sprinkling on the waste by pulses continued for 48 hours, and thereafter the irrigation reverted back to

flushing out the inherent tracer solute in the waste fill with clean water. The washout with water continued at steady state till the concentration of the tracers in the waste fill appeared to be similar to their background concentration.

Following the oversaturated tracer test, a saturation test, which has no surface overlying pond was undertaken on the waste fill. Preliminary tests had indicated that a steady-state full-saturation flow in the waste fill is obtainable at a flux of 0.547l/h; and therefore this flow rate was used in the test instead of the 1l/h used in the previous test. Prior to this test, the waste fill was completely washout of any inherent chemical compounds from the previous tests until background concentrations of the tracers in the original bare waste were established. Unlike in the previous tracer tests, automatic data logging was employed and thus only one tracer - Sodium Chloride was used for tracing the solute flow. As before, the tracer water was applied in top pulses (trickles) for 48h before being flushed out by water till background concentrations appeared to have been reached. The whole test was undertaken in 450h.

2.2 Analytical

2.2.1 Normalised solute flow concentrations

The mean values of normalised solute concentration in the mass flux at the bottom surface of the waste fill can be obtained from the existing stochastic transfer models for solute flow in a soil

medium. For a narrow pulse injection of a conservative tracer into the top surface of a waste fill under steady state flow conditions, the lifetime probability density function (pdf) of the solute travel time in the waste is log-normal and represented for any measuring depth as (Jury *et al.*, 1986; White *et al.*, 1986; Rosqvist, 1999; Rosqvist and Bendz, 1999)

$$g(t) = [(2\pi^{0.5}\sigma t)]^{-1} e^{-[\ln(t)-\mu]^2/2\sigma^2} \quad [1]$$

where t is the elapsed time since the start of flow; μ is the mean value of $\ln(t)$; and σ^2 are the standard deviation and variance of $\ln(t)$ respectively.

It has been previously proved that that physical models of solute transport that are based on the law of mass balance, when normalised are quantitatively equal to the stochastic transfer function models that are conceptualised with a probability theory (White *et al.*, 1986). Therefore, the pdf stated above can be expressed in non-parametric terms as (Jury, 1982; White *et al.*, 1986)

$$g(t) = \frac{c(t)Q}{k} \quad [2]$$

where $C(t)$ is the concentration of the solute at the depth of measurement of the waste fill; Q is the volumetric flow rate (discharge) of the solute and k is called the constant of normalisation that is equivalent to the total mass of the solute at the surface of the depth of measurement.

In general,

$$\int_0^{\infty} g(t) dt = 1 \quad [3]$$

2.2.2 Active fractional water volume in the waste fill

Ordinarily, leachate is often referred to as leachate only when it reaches the basal saturated region of the landfill. Technically, the water flowing in the waste whatever the degree of saturation or depth of landfill is a leached water which should be called "leachate" as it contains solutes consisting of waste contaminants. The active volume of water transporting solutes is usually of great interest to waste modellers and operators as it influences the operating strategy especially in accelerated biodegradation, which is influenced by the spread of nutrients, usually in solute water flow. As the microscopic flow pattern in waste is complex, quantifying the solute flux in terms of a macroscopic term such as the average solute-transporting water content in the waste fill will enhance the water recirculation process, thus early stabilization of the waste fill. The apparent fractional volume of the water within the waste fill participating in the solute transport is given as (White *et al.*, 1986)

$$\theta_{st} = q/\bar{v} \quad [4]$$

where q is the Darcian flux, and \bar{v} is the mean pore water velocity that is obtainable from the following

$$\bar{v} = L/\bar{t} \quad [5]$$

where \bar{t} is the median of the distribution of the travel time for solute particles and can be calculated from

$$\bar{t} = e^{\mu} \quad [6]$$

Alternatively, \bar{t} can be obtained analytically from the time corresponding to the 50% of the cumulative normalised flux concentration, which is easily obtained graphically.

3. Results and Discussion

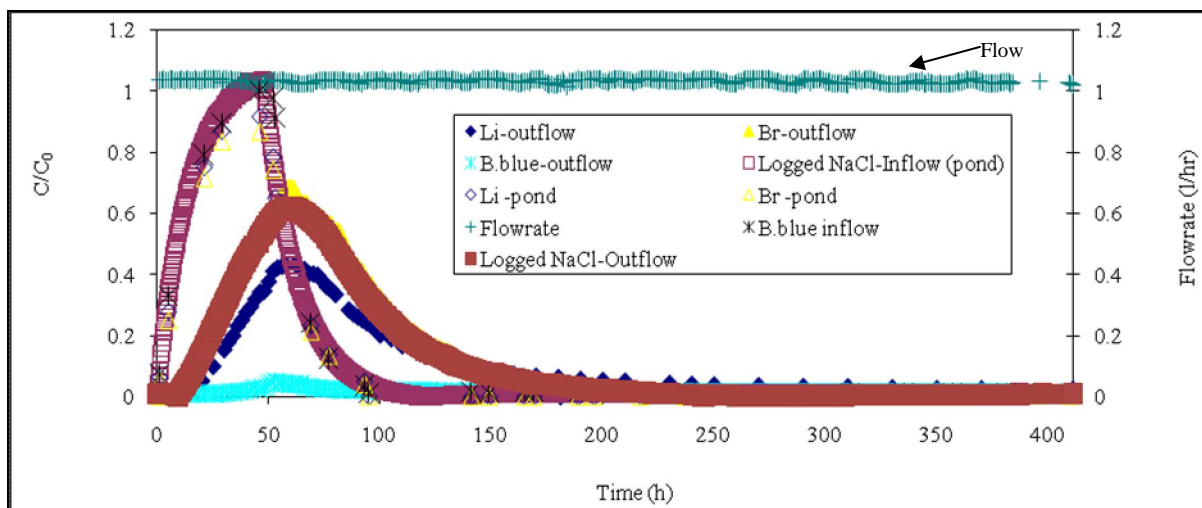
Throughout this study, the temporal solute measurements are for the exit surface of the waste fill (basal face), at the bottom of the cell, and the velocity of the solute and the bulk water motion through the waste is assumed to be the same. The hydraulic and physical properties of the waste fill during the period of the tests are summarized in Table 2. These values are similar to the properties obtained in the previous tests on the waste column, and also MSW properties obtained from large-scale test cells (Beaven, 1999; Beaven *et al.*, 2005).

The breakthrough curves (BTCs) for the oversaturated flows for different tracers and the flow rate in the waste fill is shown in Figures 2a. The BTC for the saturated flow for Sodium Chloride is depicted in Figure 2b. There is a slight difference in the shape of the washout curves for the flows but the characteristic trend for both flow conditions depicts positive skew with early peaks and long tails. These are similar to the reported typical flows that were observed for solid waste fills (Rosqvist, 1999; Rosqvist *et al.*, 2005; Beaven *et al.*, 2005). The BTC characteristics appears to be an evidence of fast preferential flow of solutes in channels and pores that are relatively larger than the average pore of the waste matrix and also an evidence of an accompanied slow flow of the solute in non preferential channels. The concentrations of the Sodium Chloride flux obtained using the parametric model (pdf) in equation 1 and the measured data, normalised using physical equations in equation 2 for the saturated and oversaturated conditions of the waste fill are depicted in Figure 3. Comparison of the model and the measured data is often necessary to determine scale dependency of the measured data (White *et al.*, 1986).

Table 2: The hydro-physico properties of the waste fill.

Waste properties	Dry density (kg/m ³)	Drainable porosity-downwards (%)	Effective field capacity (%)	Darcian flux ¹ (m/s)	Darcian flux ² (m/s)	Average linear velocity ¹ (m/s)	Average linear velocity ² (m/s)	Saturated hydraulic conductivity (m/s)
Measured values)	722	7.5	37.7	8.40 x 10 ⁻⁷	1.54 x 10 ⁻⁶	1.86 x 10 ⁻⁶	3.40 x 10 ⁻⁶	1.65 x 10 ⁻⁶

¹ – normal saturated (without surface pond); ² – saturated (with surface pond)



Li – Lithium; Br – Bromide; B. Blue – Brilliant Blue; NaCl – Sodium Chloride.

Figure 2a: The BTC for oversaturated flow in the waste fill.

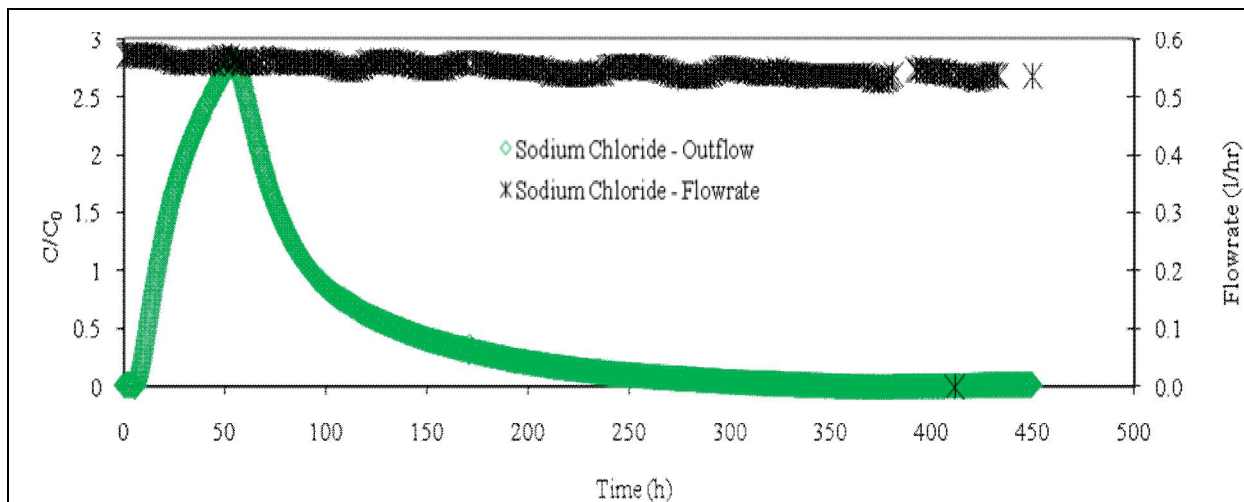


Figure 2b: The BTC for the outflow, and the flowrate during in saturated conditions of the waste fill.

The advent of computers that are equipped with fast computational programs and the robustness of Microsoft Excel program has enabled quick fitting of the model to the normalised measured data with less fitting errors. The reasonable fitting of the curves of modelled and normalised concentration of solutes thus indicates that the measured data from the test cell is not significantly affected by the scale of

experimentation. In comparison to the oversaturated condition, the normalised flow concentration in the saturated condition is slightly higher from the start of flow to the time its modal concentration is attained (Figure 3). The reason for this observation is possibly owing to the dilution of the input concentration of the tracer in the mixing pond that makes the concentration of the infiltrating tracer into the waste fill via the

surface pond to be relatively less than the concentration of the tracer source, which otherwise infiltrated directly in the saturated condition. The trend of these curves affirms previous characteristic observations on waste (Rosqvist and Bendz, 1999; Rosqvist and Bendz, 1999; Rosqvist and Destouni, 2000) - non-uniform flow owing to mechanical mixing of the solutes in fast advective water flux in preferential channels and a resultant slow solute transport owing to flow conservation and continuity in the waste fill.

The cumulative fractional concentration of the solute flow for both the model and measured data of Sodium Chloride is depicted in Figure 4. These curves are quite important as they enable the median of the pdf of the travel time (t) of solutes to be calculated graphically. It also enables the discrepancies in the model and actual data of the experiment to be visually observed. It is evident from these graphs that virtually all the recoverable concentration of the solutes has been washed out of the waste fill at the time the tests ended. The fraction of the solutes of Sodium Chloride recovered from the flux is 0.886 for the saturated flow condition and 0.918 for oversaturated flow conditions respectively. Obviously, the difference between the model and the observed data is due to the irreversible sorption of the solutes to the waste constituents, which is not filtered out in the transfer function model (equation 1). The nature or process of the sorption could be established if extensive sophisticated chemical analysis or simulation is undertaken, which is outside the scope of

this study. It could be inferred from these curves (Figure 4) that, the higher the velocity and volume of water washout, the higher the mass flush-out of solutes; which is not surprising.

The normalised flow concentration for Lithium is depicted in Figure 5. As in the case of Sodium Chloride in Figure 3, the curve fitting using the pdf model appears reasonable, however, the curve for the cumulative fractional concentration (Figure 6) shows that 0.817 of the input tracer was recoverable during the test period. The shape of the curve shows that probably further mass recovery of the Lithium solutes is possible, but the sparsely manual measured data for Lithium did not allow the tail end of the curve to be fully defined as in the case of Sodium Chloride where sufficient data obtained by the data logger enabled the plot of virtually perfect curve of the solute concentration. A good fit between the model and observed concentration data is observed for Bromide, as illustrated in Figures 7 and 8. It appears that Bromide has lesser irreversible sorption and therefore is a good conservative tracer in the saturated flow conditions of the waste. In contrast, the characteristic trend of the Coomassie Brilliant Blue flux cannot be fitted with the model at all (Figure 9). This is not surprising as significant staining and sorption of Coomassie Brilliant Blue was visually observed in the waste fill in the test cell during the tests. Less than 15% of the input mass of Coomassie Brilliant Blue was recovered from the waste at the end of the tests (Figure 10).

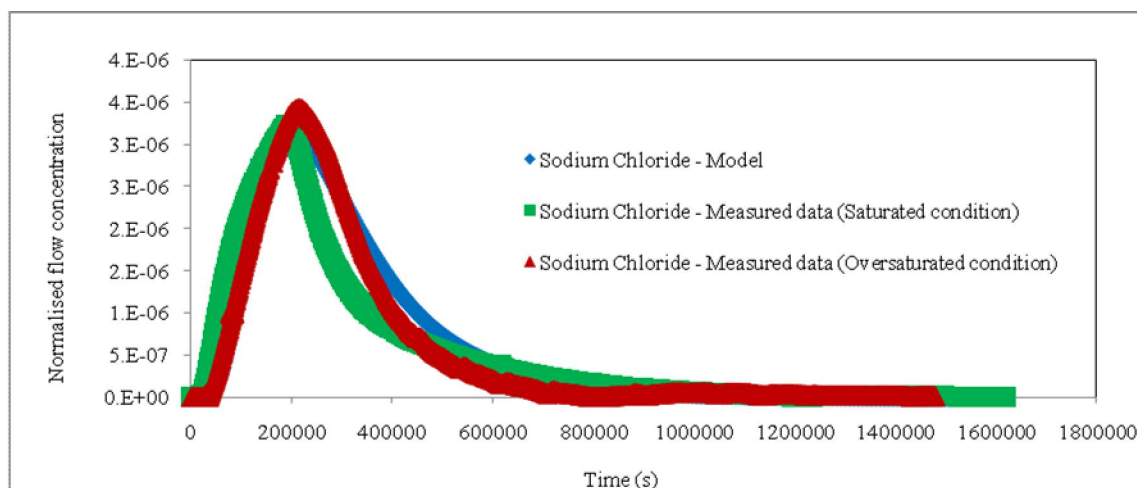


Figure 3: The normalised flow concentration for Sodium Chloride in saturated and oversaturated conditions

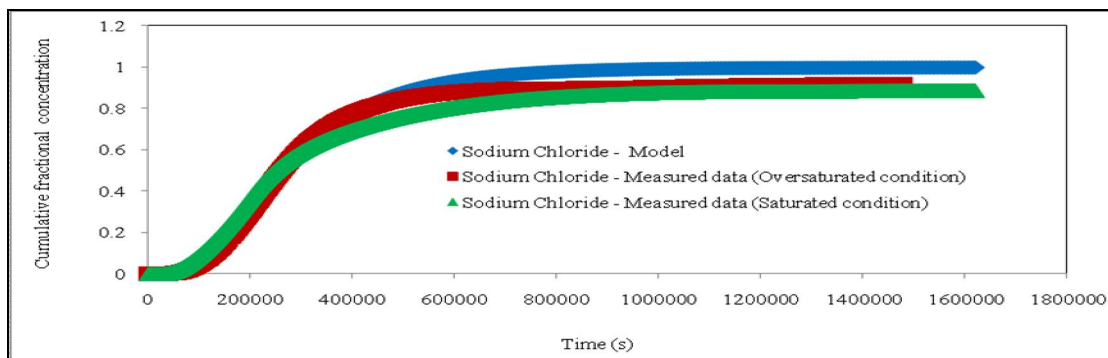


Figure 4: The cumulative fractional concentration for Sodium Chloride in saturated and oversaturated conditions.

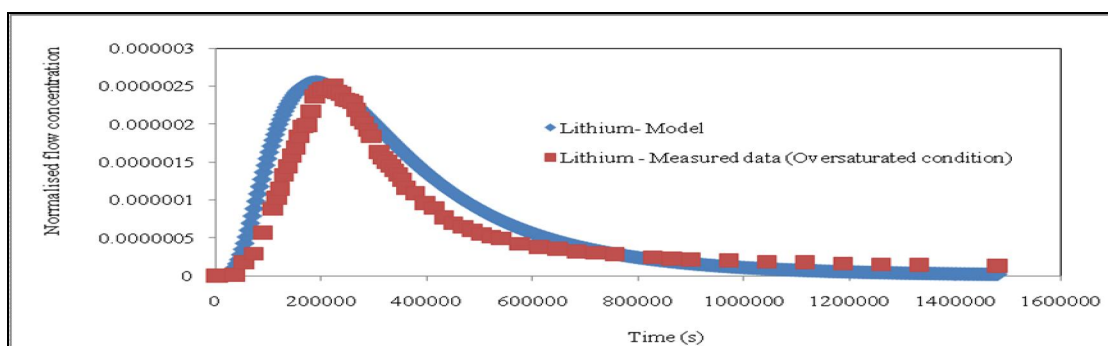


Figure 5: The normalised flow concentration for Lithium in oversaturated conditions.

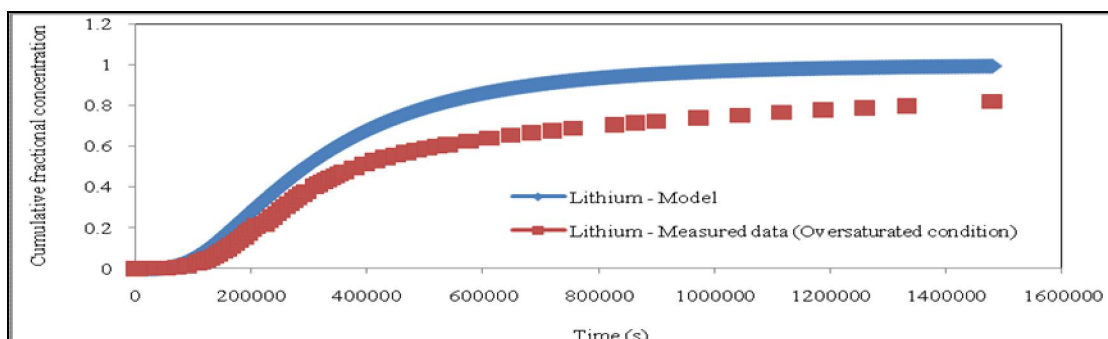


Figure 6: The cumulative fractional concentration for Lithium in oversaturated conditions.

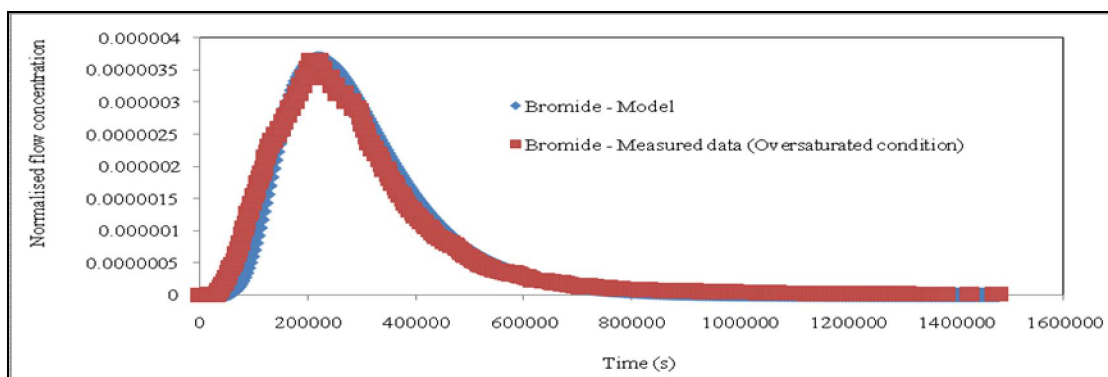


Figure 7: The normalised flow concentration for Bromide in oversaturated conditions.

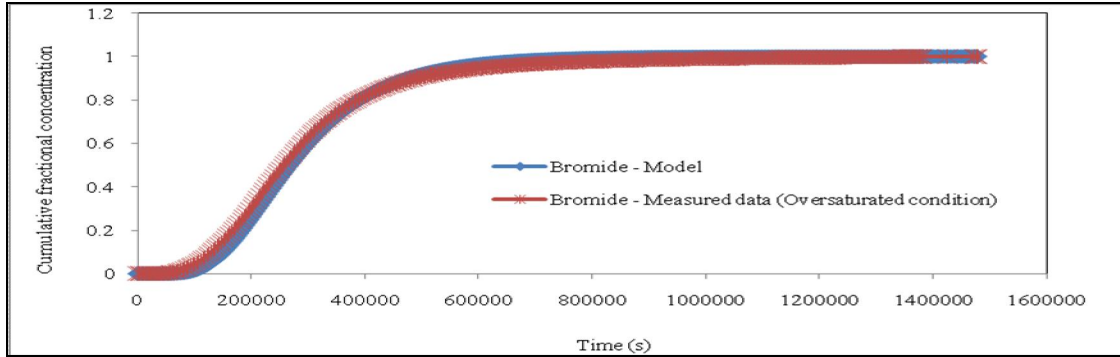


Figure 8: The cumulative fractional concentration for Bromide in oversaturated conditions.

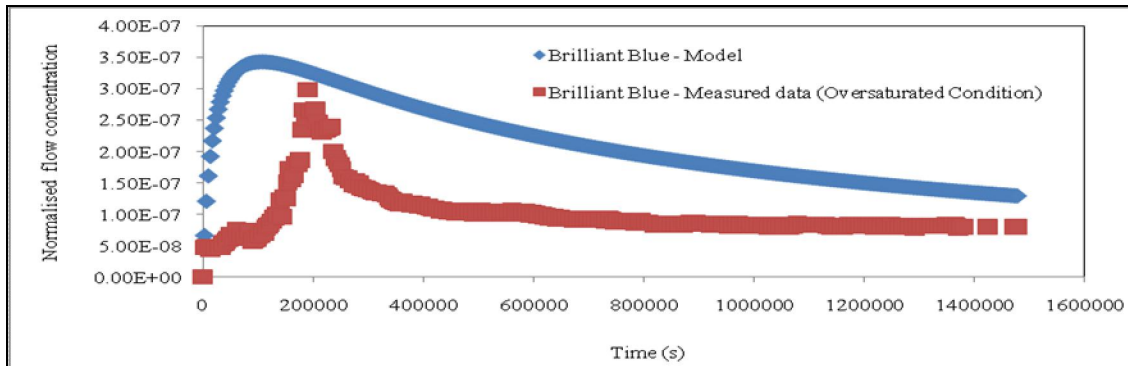


Figure 9: The normalised flow concentration for Brilliant Blue in oversaturated conditions.

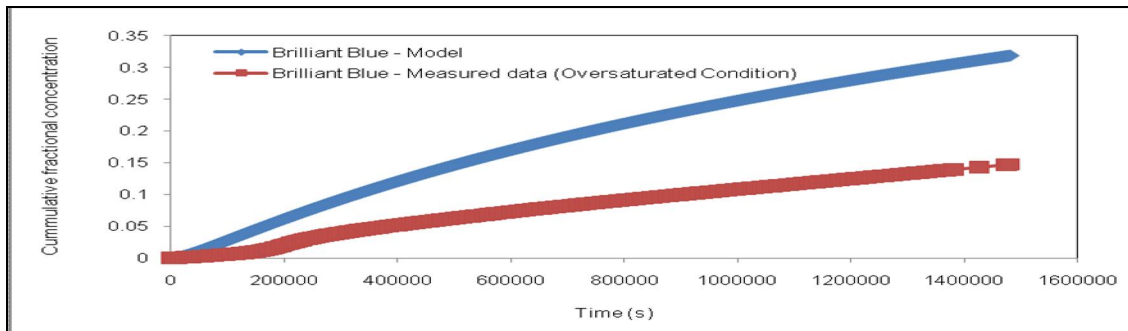


Figure 10: The cumulative fractional concentration for Brilliant Blue in oversaturated conditions.

The characteristics of the solute flux during the vertical flow of water in the waste fill is summarised in Table 3. In general, there is a reasonable agreement between the modelled and measured quantities obtained from graph, although the latter appears equal or slightly less in most instances. The unsuitability of Coomassie Brilliant Blue for tracer tests in MSW is further demonstrated with its unreasonable measured data compared with the model. The median of the distribution of the travel time for solutes (\bar{t}) has been computed using the model (equation 6), and from 50% cumulative of the fractional concentration obtained from experimental data. The median travel time for solutes (\bar{t}) calculated with the pdf model appears slightly higher than the estimation undertaken using the cumulative plot of the fractionalized measured concentration for Sodium Chloride and Bromide. Consequently, the values obtained from the latter method, which were for actual measured data were mainly used for the discussion in this study. The time corresponding to the mode of the distribution of the travel time for solutes is equivalent to the time at which the fractional modal concentration occurs (t_{modal}). This time is used to calculate the amount of fractional water contributing to the modal flow of solutes (f_{modal}) in the waste fill using a similar method as expressed in equations 4 & 5 for median travel time.

Table 3: Parameters and quantities of the fractional solute flux in the waste fill

Parameters/ Quantities	Sodium Chloride (saturated - without surface pond)	Sodium Chloride (saturated - with pond)	Lithium (saturated -with pond)	Bromide (saturated -with pond)	Brilliant Blue (saturated -with pond)
μ	12.48	12.48	12.59	12.50	15.09
	0.548	0.548	0.663	0.447	1.871
\bar{E}^i (s)	263024	263024	293608	268337	3576875
\bar{E}^b (s)	225900	249300	309600	259200	608400
t_{modal} (s)	193500	215100	223200	219600	190800
q_{st}^a (m^3m^{-3})	0.514	0.942	1.052	0.961	12.810
q_{st}^b (m^3m^{-3})	0.441	0.890	1.105	0.925	2.172
q_{modal} (m^3m^{-3})	0.378	0.768	0.797	0.784	0.681
a (%)	113.68	208.41	232.64	212.62	2834.12
b (%)	97.57	196.90	244.52	204.72	480.52
m (%)	83.58	169.89	176.28	173.44	150.69

^a - calculated from model; ^b - calculated from the cumulative graph; ^m - percentage of waste volumetric water involved in solute flux; ^m - percentage of waste volumetric water involved in modal solute flux.

Apart from Brilliant Blue, the fractional volume of the water flux participating in solute transport in oversaturated flow conditions appears approximately twice of that in the normal saturated flow conditions, as observed using the Sodium Chloride tracer. As the total porosity of the waste fill is 0.452, it appears that approximately 97.6% of the water content in the un-submerged waste fill is active in the transfer of the solutes from the inlet surface through the waste body to the basal surface, where measurements were undertaken. This indicates that the majority of the water content of the waste fill is involved in advection of solutes during the saturated flow. This further supports a previous investigation (Oni, 2009) that found a relatively enhanced water mobility in waste fills, in comparison to soils, owing to the intra-grain water flow in waste constituents.

The percentage of water participation in solute flow in the oversaturated flow condition is calculated to be more than 100% for all the tracers, even if the pond volume (11.76l) is considered in the calculation as an additional volume of water to which the waste is subjected to during the water flow. A sensible explanation of the excess percentage solute-transporting water for these tracer tests is that a significant volume of the flowing water passes through the sidewall of the waste fill. In fact, these computed values suggest that the volume of solute and water flow through the sidewall is almost equal to that within the waste body.

The volumetric water content (^m) contributing to the modal flow in saturated conditions is approximately 83.6% of the water volume in the waste fill during the flow. Thus, it can be inferred that

approximately four fifths of the water content in the waste is involved in a relative fast advective transport of the solutes, and about a fourth of this water is involved in a slower flow. The precise velocities of the varying flow regimes in the waste fill is difficult to determine but variance in the median and mode solute flux is indicated in the occurrence periods (Table 3). Similarly, the degree of the water involved in the modal flow for the oversaturated waste conditions in the cell is about twice of the un-submerged condition, which suggests that a significant proportion, approximately 50% of the modal solute flux, also pass through the side wall. This suggests that suitable side and basal containment of contaminants is needed, especially for oversaturated situations in landfills to prevent pollution of the immediate environment.

The most significant finding in this study is the active participation of almost all the water volume in a saturated MSW fill in the transport of solutes. The significant proportion of permeable constituents of waste, which enables intra-grain flow, is likely to be the reason for the enhanced mobility of water and solutes within the elements of the waste fill, as compared to soil with relatively less permeable grains.

For the fact that the majority of this water is likely to participate in the fast flow suggests that water flushing system may be effective in stabilising MWS landfills for early after-use. This study has also shown that the travel time pdf can be used to estimate the volume of the solute-transporting water, although the estimates may be slightly higher but reasonable.

4. Conclusion

This study has shown that the volumetric fraction of the water flow involved in transfer of solutes through the waste fill can be obtained from the measured temporal concentration of the effluent from the waste. The various BTCs affirm that solute transport in a waste fill is undertaken in fast advective flow domain and a relative slow flow domain within the waste matrix and grains. Insignificant mass flow of the solute owing to de-sorption seems to continue with time following stoppage of tracer input through the same transport process as in the main tracer transport.

The test results indicate that almost all the water volume of the saturated waste is actively involved in the transport of solutes through the fill. Whereas approximately four fifths of the water volume of the waste appears to contribute to the fast advective flow of the solutes, only a fifth appears to contribute to the slow flux of the solutes in the waste. The vertical flow along the sidewall in the oversaturated condition of the waste fill was as significant as the flow through the waste itself, therefore suggesting that adequate side and basal containment should be provided in cases of a submerged part of a waste landfill.

The study also indicates that the parametric model (pdf) of the travel time of solute can be used to reasonably simulate solute-transporting water in waste fills in large-scale test-cells.

Coomassie Brilliant Blue has been found to be an unsuitable tracer for flow tests in waste fills owing to its significant sorption to the waste constituents. In contrast, Sodium Chloride, Lithium and Bromide appear to be relative conservative tracers for determining the flow patterns of solutes in MSW fills.

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