

Studying Pollutant Solute Transport in Saturated MSW using Multi-Tracer Tests

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Abstract: A study of the pollutant solute flow in waste landfills has been undertaken using common tracers- Lithium, Bromide and Sodium Chloride, and Coomassie Brilliant Blue in the water flow through a municipal solid waste (MSW) fill. Vertical and recycling flow conditions of solute transport in waste fills were studied using temporal measurements of the tracers in the surface pond and the outlet flow through the waste column. The resulting breakthrough curves, which are positively skewed with long tails in various degrees, indicate non-uniformity of the gravitational flow of solute transport in the fill. Aside from the flow in the matrix of the waste fill, there is an evidence of intra-particle flow of the solute in the waste fill, which should be considered in the existing ideology of double porosity - in the storage for slow and immobile flow particles cell, and the future formulation of transfer function models for solute transport in MSW fills. There was evidence of irreversible sorption of Brilliant Blue, and also of Lithium to a slight extent. The study suggests that Bromide and Sodium Chloride that appear relatively conservative are affordable and thus appropriate for researchers studying the transport of pollutants dumped in open waste sites in developing countries.

Key words: water pollutant, preferential flow, municipal solid waste, solute transport, recycling flow, developing countries

INTRODUCTION

Pollution of the aquifers through waste landfill remains a major concern to water conservationists throughout the world. The main method of disposing solid waste in the world has been through landfill, which can be controlled or uncontrolled depending on the economy and technological advancement of the country. Controlled landfill using regulations has now become mandatory in the developed countries owing to the increasing awareness of subsurface pollution resulting from waste landfill and dumps. However, uncontrolled landfill is still prevalent in the developing countries and constitutes the largest disposal method in the world (Rushbrook, 1997; Rosqvist, 2000). Approximately 70% of municipal solid waste (MSW) generated internationally is being disposed of to landfills and about 70% of these are waste generated in households (OECD, 2009). In the UK, prior to the "Landfill Directive", the proportion of stable ash reduced while the amount of paper, rag, and plastic increased considerably owing to the 1956 Clean Air Act. Consequently, the period of time required for natural stabilisation of refuse constituents in landfills increased from years to decades, thereby limiting post-closure use (Watt, 1999). The landfill tax was introduced in October 1996 to reduce waste production by increasing the cost of waste disposal. About a fifth of the tax collected by landfill operators can be designated for pertinent projects under the Landfill Tax Credit (Waste, 1998). Among these projects are researches and techniques to achieve quick safe potential of the leachate and landfill gas produced through better understanding of the physical and biochemical processes taking place within the landfill framework.

The importance of the management of water in waste landfills has always been emphasised by various authors in achieving an acceptable low gaseous emissions and low quality of leachate (El-Fadel, Findikakis, 1997; Rosqvist, 2000; Oni, 2000; Li and Zaiss, 2001; Gawande, 2003; Rosqvist, 2005). It is common for researchers to determine hydraulic parameters such as the hydraulic conductivity of waste to enhance the efforts toward anaerobic digestion via recirculation of leachate (Chen and Chynoweth, 1995; powrie and Beaven, 1999; Oni and Richards, 2004a; powrie, 2005). Even with this, redistribution of substrates and nutrients to the anaerobic micro-organism is not optimised owing to the non-uniformity of flow in the emplaced waste

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(Rosqvist, 1999). Accurate prediction of leachate quantity and quality is a highly complex task owing to inadequate understanding of the spatial and temporal variation of the moisture in the waste fill (El-Fadel, 1997; Beaven, 2000; Oni and Richards, 2004b). As leachate comprises the solute particles of various leached contaminants, a better understanding of the solute transport in the waste fill will enhance leachate volume predictions.

The flow of water in an emplaced waste depends on the characteristics of the flow paths within the structural framework of the waste. The sources of the waste streams emplaced at landfills are heterogeneous; therefore the properties of waste within various lifts in landfills are heterogeneous. Consequently, the flow of water and solutes in waste fills are non-uniform. Numerous studies and works have been undertaken on non-uniform flow in soils. In many cases, there have been focuses on the various small and local heterogeneities causing a fast advective and non-equilibrium flow commonly called "Preferential Flow" (Jury, W.A. and L.H. Stolzy, 1982; Brusseau, M.L. and P.S.C. Rao, 1990). The theory of preferential flow was initially described by Barenblatt *et al.* (1960) to apply to seepage of liquids in fissured rocks but has since been applied to porous media containing cracks or macropores (Barenblatt, G.I., I.P. Zheltov, 1960). Preferential flow is very important for water resources sustainability as it enhances the leaching of pollutants from the ground surface to subsurface and eventually to the groundwater. A current review of preferential flow domains of the soil porous system found four distinct mechanisms at the scale of larger units of soil particles formed by aggregation (Coppola, A., M. Kutilek, 2009). These includes (i) preferential flow in real macropores, (ii) preferential flow in inter-aggregate pores, (iii) fingering due to the instability on the wetting front; and (iv) preferential flow due to spatial irregularities or temporal dynamics in soil wettability.

Waste is complex in nature; even the micro-modelling of the temporal and spatial distribution of moisture in landfills is not exact with the existing models (El-Fadel, M., A.N. Findikakis, 1997). Consequently, the majority of investigations on properties of emplaced waste have been undertaken with the assumption that the waste is homogenous (Schroeder, P.R., T.S. Dozier, 2009; Bleiker, D.E., G. Farquhar, 1995; Oni, O.A. and D.J. Richards, 2004b). This contrasts the reality, and as a result, the simulation of leachate in waste fills has in most cases not fully matched the field observations. Up till now, there have been relatively few investigations into the non-uniform flow in solid waste landfills (Rosqvist, H. and D. Bendz, 1999; Beaven, R.P., L. Dollar, 2005; Beaven, R.P., L.K. Ivanova, 2007; Rosqvist, N.H., L.H. Dollar, 2005; Rosqvist, R. and G. Destouni, 2000).

Tracer tests are often used to trace the path of flowing water. Ideally, tracer chemicals must behave conservatively, with the only transport processes influencing a conservative tracer being advection, dispersion and diffusion. The most common tracers used in groundwater are fluorescent dyes and halides such as chlorides, bromides and iodide. Dyes have the ability to stain the travel paths of water and solutes in soils and are thus useful for visualizing water flow patterns in any medium. For instance, Coomassie Brilliant Blue dye has a solubility of 200 kg m⁻³ and is particularly suited for field use as it is nontoxic; depending on pH, the dye is either neutral or dissociates to a mono- or bivalent anion (Flury, M. and H. Flühler, 1995). In the developing countries, Sodium Chloride and the Coomassie Brilliant Blue dye are readily available and therefore may be suited for tracer tests to determine preliminary studies of the likeable degree of contamination of a pollutant source. In fact, Coomassie Brilliant Dyes are a family of dyes commonly used to stain proteins in sodium dodecyl sulfate and blue native polyacrylamide gel electrophoresis. The original Coomassie dye was developed as a wool dye and named to commemorate the 1896 British occupation of Coomassie (now Kumasi) in Ghana, West Africa. In landfills, Lithium is the most popular tracer of landfill leachate owing to its usually undetectable quantities and minimal interaction with the waste (Öman, C. and H. Rosqvist, 1999; Pocachard, J., 2005).

This study has been undertaken to fully understand the flow pattern of dissolved pollutants in an emplaced waste fill so as to enhance predictions of leachate from landfills. Multiple tracer tests have been undertaken in a saturated, submerged condition of a waste fill. Four common tracers, Lithium, Bromide and Sodium Chloride, and Coomassie Brilliant Blue were used to study the preferential flow in a waste column consisting of well-degraded waste constituents. The concentration of Sodium Chloride as a compound other than as ions of Na and Cl were measured indirectly from the EC using the Campbell Scientific's datalogger and the manual "Hanna" probe.

MATERIALS AND METHOD

Experimental Set-up:

The cell used for undertaking tracer tests on the waste comprised a cast acrylic cell, 0.480m inner diameter, 0.90m high, and 12mm wall thickness mounted in an ELAND structure (Figure 1). The ELAND

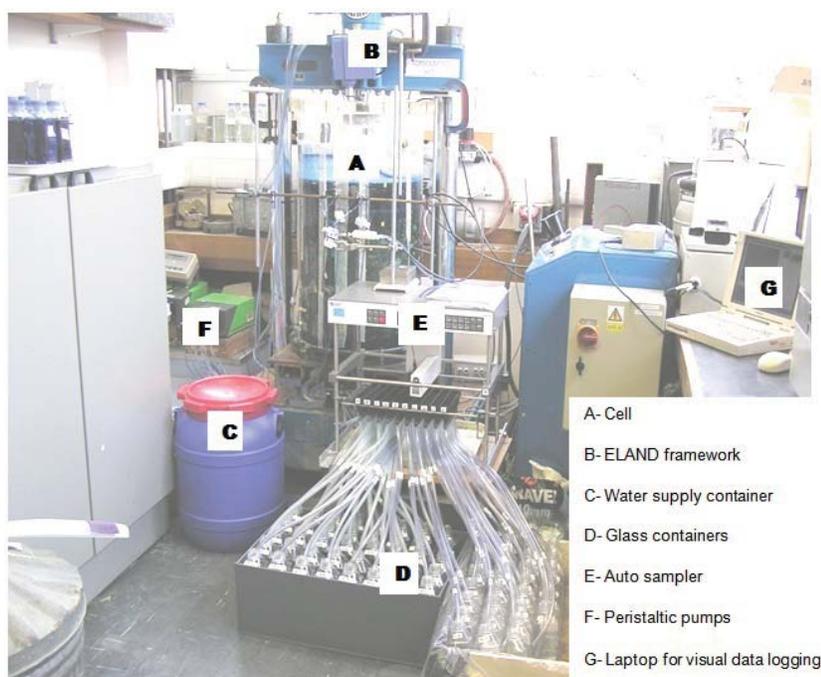


Fig. 1: Experimental set-up

framework is capable of applying a constant stress to the waste column in the cell through a top platen on the upper surface of the waste. The test cell had been previously used as a Consolidating Anaerobic Reactor (CAR) for MSW materials (Ivanova, L., 2003). The cell however in this study has been improvised to be suitable for tracer tests in a waste column. Radial holes, 12mm diameter were drilled uniformly on the top platen to enable inflow of water and tracer at the surface of the waste. It also enables vertical flow under gravity through the waste. The top platen, stood 50mm clear of the top of the cell to enable sampling during the tests. A circular galvanised iron mesh was placed on the surface of the waste fill to prevent vertical movement of lightweight particles during the flow of water in the cell. There were three ports located at the side of the cell at approximately 100mm, 250mm, and 400mm above the base of the cell respectively. Vertical tubes attached to these ports served as manometers to measure the hydraulic head. In addition to this, eight ports uniformly located at the base of the cell served as outflow channels of water from the sample.

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 ports was conveyed through a 3mm diameter silicon tube, 1m long, connected to a lower end of an inverted Y piece-connector, fastened to an elevated horizontal bar. The bar was firmly fixed onto two vertical metal stands. The other lower end of the Y piece-connector was connected to a 3mm-diameter tube, 90 cm long, which was further connected to an automatic sampling collector. The top end of the Y connectors was open to enable atmospheric pressure on the outflow, and has a free water surface 3.5 cm below the surface of the waste fill. For data logging of the experiment, an inlet probe was put on the top of the waste fill in the surface pond and an outlet probe was attached in between the outlet port of the cell and the auto-sampler. In order to prevent overflow in the inverted Y piece-connector as a result of entrapped air in the system, the common outlet tube was made to discharge to a fabricated “stilling well” creating an atmospheric break. A 3mm diameter silicon tube connected the discharge from this “stilling well” to the outlet probe. The whole unit was fixed with load cells to enable the monitoring of temporal mass and therefore volume of the fluid in the cell. Also, an electric stirrer was secured to the frame of the ELAND for uniform mixing of the surface pond in the cell. Water was pumped into the cell initially from a 20-litre water bucket, but later replaced with a 50 litre container with two peristaltic pumps (Figure 1) when the actual tests started. The pumps, which initially had been calibrated, were expected to complement each during an unforeseen breakdown of any of them.

Test Material:

The waste materials used in these tests have been previously described (Ivanova, L., 2003). It comprises a 15-year-old solid waste sample obtained from a MSW landfill site in Rainham, Essex, UK. Prior to infilling, the waste was shredded to maximum particle size of 40-50cm and then reconstituted to a composition characteristic of the landfill (Table 1). Initially, the CAR 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 emplaced height of the waste decreased from 0.623m to 0.572m equating to an increase in dry density from 498kg/m³ to 572kg/m³.

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

Test Procedure:

Prior to the tracer tests the waste had been flushed out of the bio-chemical residue of the previous tests by clean water and allowed to drain for a long period of time. An applied load of 150kPa was applied constantly to the waste, which overlays approximately 0.1m layer of 10mm washed river gravel in the cell until no rebound was noticed. At this period, the height of the waste column in the cell decreased to 0.43 m. The waste was saturated via the water distribution system and then drained sequentially from the bottom of the cell. This procedure was repeated four times to minimise the entrapped air in the waste, which might inhibit the gravitational vertical flow through the waste during the tracer test. Initial chemical analysis of the gas in the outflow from waste indicated merely entrapped air. This is expected as the waste is old and well degraded and also as a result of the complete wash-out from the previous experiment. Earlier, field capacity of the waste fill was determined from the volume of water absorbed in the dry particles before any gravitational flow.

The porosity and saturated hydraulic conductivity of the waste fill were undertaken prior to the tracer tests. As before, the waste was saturated sequentially, and water levels in the manometer tubes at each water increment were measured as soon as the levels stabilised. This process was repeated with a reverse in the direction of water flow so as to obtain the porosity in both vertical directions of flow. Then, water was recirculated in the cell at a flow rate of 1l/h until a steady flow was established. Following this, the hydraulic gradient and the volume of surface pond water required for gravitational vertical flow through the waste fill were calculated. The saturated hydraulic conductivity was calculated from the measured hydraulic gradient and flowrate using the Darcy's law. Recirculation of the leached water continued prior to the tracer test.

The four tracers, Lithium (30mg/l), Bromide (345mg/l), Coomassie Brilliant Blue (0.1g/) and Common salt (5g/l) were vigorously mixed water in a 50l-container until total dilution was achieved. Six samples were taken from the bulk solution to ensure the designed concentrations were obtained. Approximately 48h prior to the start of tracer test, the water recirculation was stopped and vertical flow was established with inflow into the surface pond on the waste fill through the water distribution system. Unlike previous tests undertaken on the cell (Beaven, R.P., L. Dollar, 2005), 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. Shortly before the start of the tracer test, four samples of the outflow from the waste were collected in 25ml bottles to establish the background concentration of the tracers in the waste fill.

At the start of the experiment, the peristaltic pump tubes were removed from the water supply container and immediately inserted into the 50l-container of tracers for continuous sprinkling of the tracer in "top cap" pulses in the cell at 1l/h to maintain the already established steady flow. At every hour, the auto-sampler collects the effluent from the waste fill in 1 litre glass, of which sub- samples for tracer analysis were obtained. Tracer irrigation of the waste fill by the pulses continued for 48 hours before switching back to water for washout of the inherent tracers. Continuous sprinkling of clean water at 1l/h continued for the next 363h for washout of the remaining tracer in the waste fill. The flow network was then shut for 72h. Following this, water recycling continued for 215.25h to enhance steady state conditions prior to the tracer recycling tests. For the tracer recycling tests, the surface water pond on the waste fill was pumped out and an equal volume of tracers was immediately pumped into the cell. In this case, the tracers used were Lithium, Sodium Chloride and Bromide. Owing to the experience in the vertical tests, a continued use of Coomassie Brilliant Blue as a tracer was found untenable. This process of instant inputting of tracer into the surface pond was undertaken within 5 minutes. A recycling flow of the effluent from the waste fill (water and solute particles) then commenced forming a "closed circuit of solute transport" and resulting in the mass conservation of the applied

tracers in the waste fill. This process continued till a time when equilibrium of the inlet (pond) and outlet concentrations of the tracer in the waste fill respectively was deemed to have been sufficiently established. The measurement of Bromide was done with an ion selective electrode while the analysis of Lithium was undertaken using an Atomic Absorption Spectrophotometer.

Temporal Solute Concentration:

As this study is different from the typical solute transport in unsaturated conditions in a waste fill, it was desired to model the solute concentration in the surface pond with time to be able to simulate the expected solute mass flow from the waste fill, assuming uniform flow conditions. The equation for the temporal concentration of the surface pond was derived from the law of mass balance and the principle of flow continuity as applied to a constant-volume pond, subjected to an incremental and elemental solute concentration in steady state conditions. The temporal concentration in the pond is thus modelled from the following equation:

$$C_i = \left(\frac{1}{V}\right) \times \left((M_{i-1} + \Delta M) - \left(\left(\frac{M_{i-1} + \Delta M}{V + \Delta V}\right) \times \Delta V \right) \right) \tag{1}$$

where:

C_i = concentration of the tracer (solute) in the pond at time i

V = constant total volume of the pond

M_{i-1} = the mass of the tracer (solute) in the pond at time $i-1$

ΔM = incremental mass of the tracer (solute) added to the pond from time $i-1$ to time i

ΔV = incremental volume of the tracer (solute) added to the pond from time $i-1$ to time i

RESULTS AND DISCUSSION

The properties of the waste fill in the cell are given in Table 2. These values are typical of MSW properties obtained from large-scale test cells (Beaven, R.P., L. Dollar, 2005). All the measurements for the vertical flow of the tracers through the waste are deemed the properties of the waste fill at the lower surface overlying the basal gravel. This is reasonable as the free drainable porosity of the gravel far exceeds that of the waste, therefore the flow through the waste pass unhindered through basal gravel.

Owing to the heterogeneity of waste, the porosity was determined in both vertical directions. For a more precise determination of the vertical porosity, the hydraulic head measured in the vertical manometer tubes attached to the side of the test cell is plotted against the volume of the leachate drained from the waste and the basal gravel (Figures 2 & 3). The porosity of waste and the basal gravel from Figure 2 is calculated for upward vertical flow as below:

$$\text{Drainable porosity}_{(waste)} = \frac{100}{63.11 \times 3.142 \times 0.24^2} = 8.8\% \tag{2}$$

$$\text{Drainable porosity}_{(gravel)} = \frac{100}{18 \times 3.142 \times 0.24^2} = 30.7\% \tag{3}$$

Similarly, the porosity of waste and the basal gravel from the above Figure 3 is calculated for downward vertical flow as below:

$$\text{Drainable porosity}_{(waste)} = \frac{100}{73.85 \times 3.142 \times 0.24^2} = 7.5\% \tag{4}$$

$$\text{Drainable porosity}_{(gravel)} = \frac{100}{18 \times 3.142 \times 0.24^2} = 30.7\% \tag{5}$$

The gradient of these plots give the inverse of the rate of volume per unit height of all the drainable pores in the vertical direction. The drainable porosity is taken as the matrix porosity excluding the small gas pockets within the waste. The drainable porosity is calculated as the ratio of the volume per unit height of the drainable pores to that of the total volume per unit height of the waste fill. This method is faster and more accurate than the traditional methods. It could be seen from these plots that the porosity determined for the waste in the downward direction is less than in the upward direction. This is not surprising since air pockets existing within the waste particulate will reduce the effective porosity and this entrapped air is easily displaced with rising water from beneath other than falling water from above, under atmospheric pressure. The large discrepancy in the field capacity and the drainable porosity shows a high percentage of the water is held within the fabrics of the waste particles. Similarly, the high value of the field capacity indicates high content of absorbent particles in the waste. This is not surprising as the classification of the degraded 15-year old MSW indicates high content of paper, wood, textile and other paper products (Table 1). Using the bed volume, the volume of the drainable porosity of the waste fill was calculated as 5.84l. The volume of the overlying surface pond was calculated as 11.76l.

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

Waste properties	Waste thickness	Dry Mass (kg)	Dry density (kg/m ³)	Drainable porosity-downwards (%)	Effective field capacity (%)	Darcian flux (m/s)	Average linear velocity (m/s)	Saturated hydraulic conductivity (m/s)
Measured values)	0.43m	~56	722	7.5	37.7	1.54 x 10 ⁻⁶	3.40 x 10 ⁻⁶	1.65 x 10 ⁻⁶

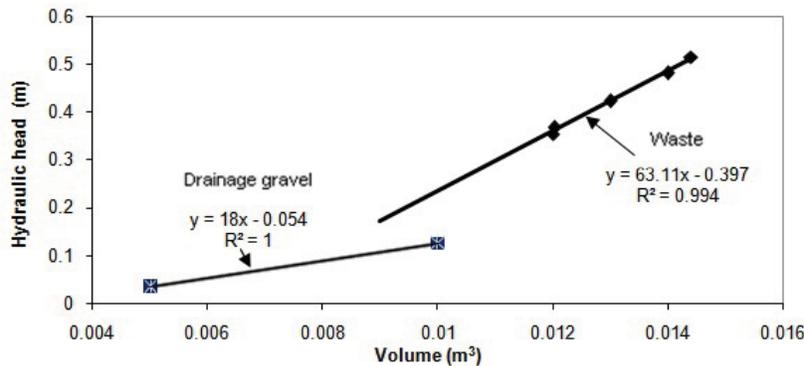


Fig. 2: Determination of vertical porosity (drainable) from upward flow

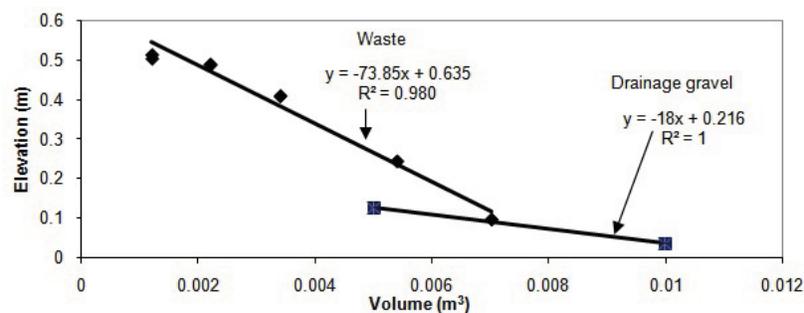


Fig. 3: Determination of vertical porosity (drainable) from downward flow

Although the manual and logged EC values of Sodium Chloride in the test are similar, the logged data has been used throughout the study owing to its larger dataset (data taken every 15min) and reliability of the measuring instrument. The concentration of the tracers within the first 15h is shown in Table 3. Assuming a uniform flow in the waste fill, the first plug trace of the tracers should appear from the basal face of the waste fill at approximately 35.17h. However, it could be seen that following 10 h, measurable concentrations of the tracers were observed. It appears that the “advective” solute particles of Bromide and Sodium Chloride have the highest linear velocity among the tracers within this period.

The breakthrough curves (BTCs) for the tracers in terms of chemical concentrations are shown in Figures 4-7. If the flow is uniform and there is no mass loss of solute in the waste fill, the BTC for each of the output tracer will be similar to the input tracer, however, with an offset of 35.17h from the start of the experiment. The maximum fractional concentrations of Lithium, Sodium Chloride, Coomassie Brilliant Blue and Bromide at the waste basal surface are 0.434, 0.621, 0.052 and 0.684 respectively, quite short of the calculated value of 0.98, for a uniform flow through the waste, with mass conservation of the solute flux. Within the period of tracer injection, the maximum fractional concentrations of the tracer in the pond are approximately 1 for Sodium Chloride and Coomassie Brilliant Blue and approximately 0.9 for Lithium and Bromide respectively (Table 4). Ordinarily, the dilution of the input tracers with the initial water content in the pond is expected and the concentrations of the tracers in the pond are expected to increase with the tracer input until either a maximum is achieved or the design concentrations of the tracers are attained prior to the water washout. Comparison of the observed concentrations with the calculated maximum fractional concentration of the tracer in the pond (0.98) indicates that the mixing in the surface pond is reasonable and the solutes particles of the tracers are not precipitated on the surface of the waste fill. The BTC curves are positively skewed, indicating non-uniformity in the flow of solutes in the waste fill. Generally, the early peaks indicate rapid flow in preferential flow paths or domain and the prolonged tails suggests there is a slow flow of solute transfer from a less mobile domain or/and exchange of diffusive mass between the two varying flow domains. These domains can be the fast flow domain and “dynamic stores” within the waste fill. A good visualisation of the characteristic trend of the solute transport in the experiments is further demonstrated by expressing the observed data in terms of the fraction of measured concentration in waste to input concentration (C/C_0), as delineated in Figure 8 for Sodium Chloride. It could be seen that the calculated and observed fractional concentration for the surface pond are very similar. However, the observed and the calculated values of the temporal fractional concentration of the effluent tracer assuming a uniform flow are dissimilar.

Table 3: Temporal fractional concentration of the tracers

Elapsed Time (h)	Lithium C/C_0	Sodium Chloride C/C_0	B.Blue C/C_0	Bromide C/C_0
10	0	0.012	0.01	0.012
15	0.031	0.067	0.012	0.064

Table 4: Useful tracer characteristics (vertical flow only)

Quantities	Lithium	Sodium Chloride	B.Blue	Bromide
Modal concentration in pond; mg/l	28	5000	100	300
Modal concentration in waste; mg/l	13.02	310.37	5.15	236.22
Elapsed time for modal concentration in pond; h	48	48	48	48
Elapsed time for modal concentration from waste; h	62	61	53	61
Calculated time for modal concentration from waste; h	83.17	83.17	83.17	83.17
Elapsed time for actual median concentration from waste; h	~86	~69	~169	~72
Calculated time for median concentration from waste; h	70.8	70.8	70.8	70.8
Equilibrium concentration (calculated; mg/l)	7.52	1252.70	25.06	86.44
Equilibrium concentration (calculated; C/C_0)	0.25	0.25	0.25	0.25
Washout (%)	~82	~92	~15	~100

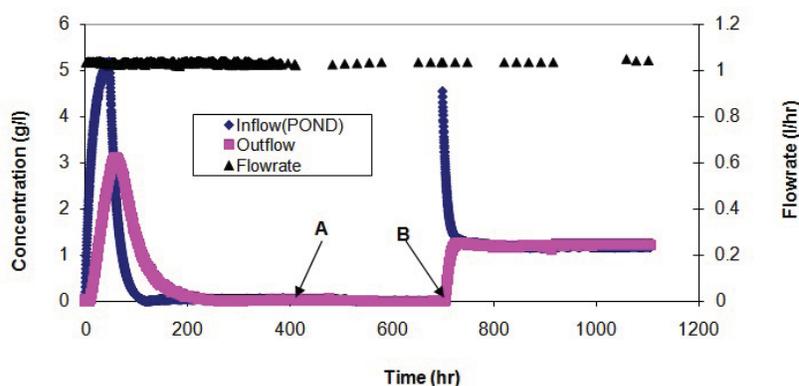


Fig. 4: Breakthrough curves for Sodium Chloride (logged data)

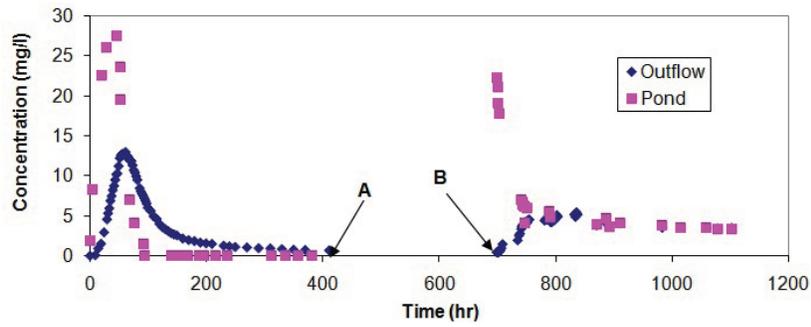


Fig. 5: Breakthrough curves for Lithium

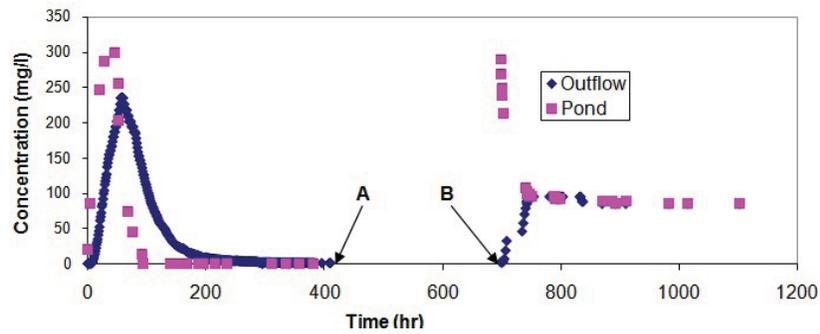


Fig. 6: Breakthrough curves for Bromide

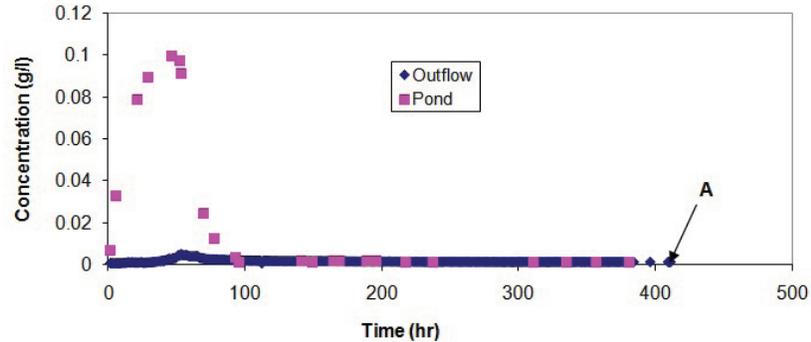


Fig. 7: Breakthrough curves for Brilliant Blue

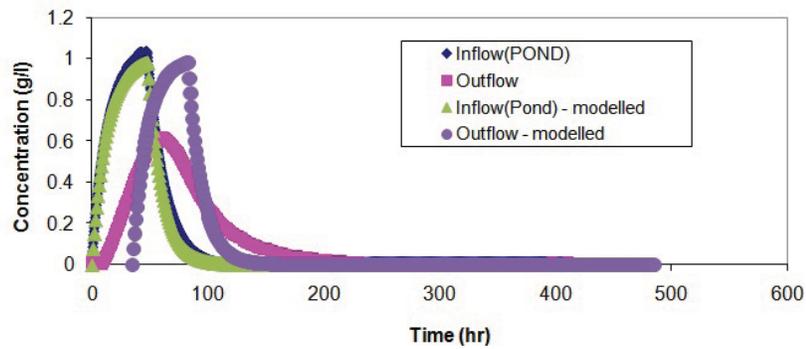


Fig. 8: Calculated and observed fractional concentration for Sodium Chloride

The types of trend curves of solute transport observed in this study have been simulated indirectly with a high degree of accuracy using a two-parameter log-normal probability distribution function (pdf) and its parallel non-parametric function (Jury, W.A., 1982; White, R.E., 1987). The fitting parameters have been useful in describing a transfer function concept in which there is a mobile fraction and an immobile fraction of the water-containing solute in a porous media. This concept has allowed the solute transport volume fraction for modal concentration flux and median concentration flux to be calculated [24, 15]. Similarly, this concept is often described analytically with the concept of double porosity in which the waste particulate consists of blocks of constituent materials containing an immobile water fraction separated by fissures through which a mobile water fraction flows (Beaven, R.P. and A.P. Hudson, 2002; Beaven, R.P. and A.P. Hudson, 2003). It is assumed that aside from the fissures, there is no advective flow in the matrix of the waste but diffusion according to Ficks Law occurs between the matrix and the adjoining fissures. The characteristics of double porosity model based on this theory is greatly influenced by (i) the ratio of mobile to immobile porosity, (ii) the time required for an elemental volume of the mobile water to reach diffusive equilibrium with equal volume of neighbouring immobile water and (iii) the advection time, which is the time required for water in the mobile fraction to move from the point of introduction into the flow fissures to the point of measurement (Wright, T.E.J. and J.A. Barker, 2001). However, a recent study suggests that modelling of the flow through waste using a probabilistic Langragian model is more accurate and realistic and that the majority of models that are based on local dispersion or local storage away from preferential channel are less realistic in explaining the solute flow in a waste fill (Rosqvist, R. and G. Destouni, 2000). It is believed that the observed flow in waste could only be simulated realistically by a two-domain conceptual model, assuming mobile water and advective solute transport both through the preferential flow paths and zones of slow flow.

In trying to further understand the mass flow of solute in the waste fill, the cumulative mass of the solute flux was calculated by summing each individual mass flux calculated for a time period of 1h using the trapezoidal rule. The actual time for the median (or average) mass flux was obtained from the time at which 50% of the cumulative mass flux exited from the basal waste surface (measuring line). This is easily obtained from the graph of cumulative mass flow shown in Figure 9. Theoretically, the time required for the median solute mass of the tracers to pass through the waste for a uniform flow is approximately 59h for unsaturated condition and approximately 71hrs for saturated “ponding” conditions (Figure 10), as in this situation. These values are summarised in Table 4. It could be seen that the median times for the mass tracer flux to pass through the basal surface (72h - 86h) are relatively similar to the calculated values (70.8h) for the tracers except for Coomassie Brilliant Blue, whose time is approximately twice the others. Owing to flow continuity in the waste fill, It can be inferred that even though there are faster advective solutes than the average linear velocity of flow, there also exist slower solutes than the average linear velocity of flow that thus make the average mass flow of the solutes to be similar to that for a uniform flowrate in the waste fill. This notion of the fast advection of solute transport is demonstrated in the time at which modal concentrations are reached in the waste fill. The observed time at which the modal concentrations of the tracers from waste were reached is more than 20h less of the calculated time for attaining this modal concentration, if the flow has been uniform. In fact, the time required for modal concentrations to be achieved at the effluent solute of Coomassie Brilliant Blue is 8h less than in the other tracers. Further evidence of fast advective transport of solute is shown in the elapsed time between modal concentration in the surface pond and in the basal surface of the waste. Using the average linear velocity for a uniform flow in the waste, the calculated time of approximately 35h is more than double the observed times for the tracers (Table 4).

The washout percentage of the tracers shown in Table 4 should be taken as the extreme maximum values. The true values probably lie between 5%-10% of these values owing to factors, including the error in the equipment used for measuring concentration and the trapezoidal method of estimating the temporal mass flow. Aside from Coomassie Brilliant Blue, a significant mass of the tracers was washed out of the waste. The mass recovery of Coomassie Brilliant Blue is just over one tenth. Possible explanation for this is a permanent staining or irreversible sorption of the Coomassie Brilliant Blue compared to the other tracers. This could be explained in terms of a relative physico- chemical attachment or absorption of its solute to the waste particles. Consequently, the solute mass flow of the Coomassie Brilliant Blue will be retarded, resulting in reduction in the fractional concentration of the Coomassie Brilliant Blue that passes through the fast preferential flow channel (Figure 7). As the optimal concentration of the Coomassie Brilliant Blue is reduced, time required for the attainment of the modal concentration in the waste fill is thus reduced, as noted above.

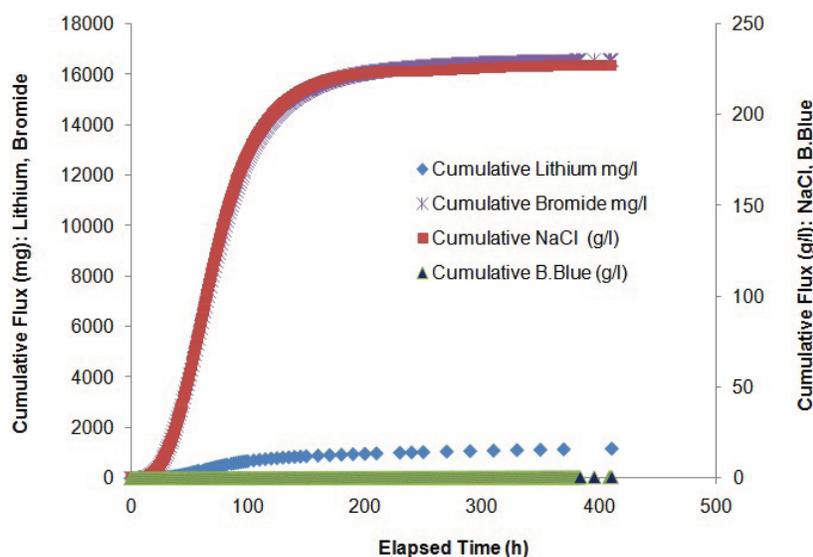


Fig. 9: Temporal cumulative mass solute flow from the waste

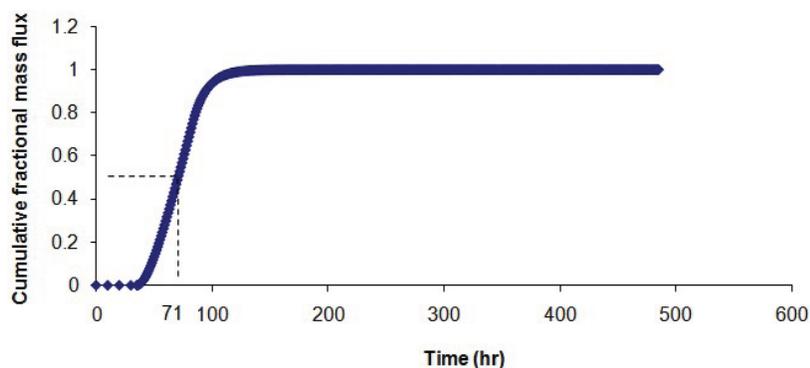


Fig. 10: Cumulative fractional mass solute flow from the waste

The non-conservativeness of the Coomassie Brilliant Blue tracer is probably enhanced with the relative slow volumetric flow through the waste, which enabled sorption and absorption of its solute particles with highly absorbent materials in the waste fill, as compared to soil that lack such materials. Coomassie Brilliant Blue tracer has been successfully used as tracers in groundwater contamination/pollution studies (Flury, M. and H. Flühler, 1995) as adsorption of Brilliant Blue is greatly reduced in soils at high velocity.

A tracer recycle in the waste fill was undertaken until a period when equilibrium concentration of the tracer solutes appeared to have been attained within the waste fill. This was to make sure that the equilibrium concentration initially attained was not transient and also to observe the trend of long-time characteristics of solute sorption in the waste. The BTC for the tracer recycle tests of the tracers is depicted in right section of Figures 4-7, starting from an elapsed time of 698.25h since the beginning of the entire experiments. The start of the period where equilibrium concentration appears to be attained for the tracers appears to be 170h following the start of the tracer recycling test. It could be seen from Figures 4-6 that while the Bromide and Sodium Chloride concentrations at equilibrium are similar to calculated values, the equilibrium concentration of Li observed deviates by half of the supposedly value. Possible explanations for this behaviour are (i) irreversible sorption of the Lithium to the waste constituents, and (ii) precipitation of the Lithium in the leachate/tracer solution. The magnitude of these could only be determined if highly complex bio-chemical analysis of the tracer in an environment similar to the waste fill are carried out. The success of such tests remains uncertain as many tests involving waste, owing to the extremely heterogeneity in “colour and character” of waste.

Tables 5a & 5b show some relevant properties of the recycled solute transport. Comparison of the calculated equilibrium concentrations in Tables 5a & 5b with the observed values in Figures 4-6 indicate that the solute does not only flow through the drainable pores only but also through the intra-particle pores of the waste. The equilibrium concentration is at least two and a half multiples of the observed values (Table 5b) if only the matrix pore is involved in the dilution of the solute in the waste. This observation is very significant as intra-particle flow does not occur in porous soil media. It thus implies that intra-particle flow of solute should be included in the assumptions and considerations when formulating waste flow models, which have been based on the fundamental principles of soil mechanics. Furthermore, the equilibrium concentration of a conservative tracer once obtained through a recycling flow pattern as this test could be used to calculate the total effective porosity of a waste fill. It thus means the average linear velocity and other parameters that are influenced by porosity, which are used to calibrate models used in simulating moisture flow in waste could be accurately estimated as well.

Table 5a: Useful tracer characteristics for recycling flow (total waste pore consideration)

Parameters	Lithium	Sodium Chloride	B.Blue	Bromide
Total pore volume + pond (l)	46.94	46.94	46.94	46.94
Recycling mass (mg)	352.8	58800.0	1176	4057.2
Calculated equilibrium concentration (mg/l)	7.52	1252.70	25.05	86.44
Calculated equilibrium C/C_0	0.251	0.251	0.251	0.251

Table 5b: Useful tracer characteristics for recycling flow (drainable waste pore consideration)

Parameters	Lithium	Sodium Chloride	B.Blue	Bromide
Drainable volume + pond (l)	17.60	17.603	17.60	17.60
Recycling mass (mg)	352.8	58800	1176	4057.2
Calculated equilibrium concentration (mg/l)	20.05	3340.87	66.82	230.52
Calculated equilibrium C/C_0	0.668	0.668	0.668	0.668

The interesting observation in the recycling tracer test is the temporal sorption of Lithium solutes which has not been definitely revealed in the vertical tests. This shows that the recycling of the effluent enables more chemical interaction of the solute and waste constituents than in just vertical flush-out from the waste fill. It is reasonable to say that the application of the properties of waste obtained from the recycling tests to the operational strategy of recycling bio-reactor would yield better results than from a vertical test. Some other advantages as observed from the findings from a similar test, though with different waste densities and tracer types (Beaven, R.P., L.K. Ivanova, 2007) are (i) the test is less laborious compared to traditional tests as it does not require attendance if manual sampling is not required, (ii) It is fast and accurate as the resultant concentration need not be zero (iii) Any error neglected as a result of the tailing-off of the BTC does not exist (iv) It is applicable to close-loop bio-reactors (v) conservation and equilibrium of the hydraulic and physico-chemical properties of the bulk waste is achieved. Tracer tests in a similar “close circuit” flow in solid waste fill has been preliminarily modelled

An interesting observation in this study is the shape of the BTC for the tracers in the pond and from the tracer exiting the waste. The shape of the BTC for the pond is similar to the BTC often observed for unsaturated flow through a waste. A likely explanation is that a larger portion of solute flow in unsaturated waste moves through the preferential channel, therefore, as soon as the tracer input is stopped, there is a marked decrease in the concentration of the solute within a short time. In this case, the contribution of the solute flux from the secondary store or less mobile element of the waste to the preferential channel or mobile element is small compared to the initial contribution from the direct tracer input. In contrary, a larger portion of solute in the submerged saturated waste is retained in the immobile/slow store that probably includes the fabrics of the waste particles. Therefore, a sudden stoppage of the tracer input is not immediately reflected in the concentration of the output tracer as part of the tracer within the fabrics are released back to the matrix porosity, and eventually into the preferential channel for advective flow through the waste. Another reason is that the concentration of the surface pond does not immediately become zero as the tracer input is stopped; it decreases gradually owing to increasing dilution by the flush water. These two processes probably explain the smooth symmetric crest of the BTC for the submerged flow in the waste.fill.

In general, the observed characteristics of the tracers used in this study have advanced the previous knowledge on the water flow in waste materials, which is different from soil. Any modeller of leachate generated or emitted from waste fills should henceforth consider the enhanced matrix flow owing to the distinct characteristic of inter-particle flow existing within the waste framework, compared to soil. By doing this, the prediction of potential leachate volume generated in landfills will be improved and adequate measures will be taken to provide basal leachate collection that will prevent pollutant migration from the landfills to the

immediate environment. In the current tests, great similarities in the characteristics of the Bromide and Sodium Chloride have been observed. In most cases these two tracers appeared to have behaved conservatively and may be used to effectively determine the flow properties of a MSW fill. The concentration of these tracers are easily determined using calibration curves plotted for the signal data measured by cheap and reliable probes. Consequently, these tracers appeared to be well suited for researchers in the developing countries who are committed to sustainable development. The use of Coomassie Brilliant Blue for dyeing textile is prevalent in the majority of African countries. However, the use of this dye as a tracer is only suited for groundwater studies owing to the high content of absorbent materials in the waste disposed of to waste dumps in developing countries. In the majority of African countries, the practice of solid waste recycling is barely in-existence, and therefore chunks of paper and absorbent materials are common in the waste dumps.

Conclusion:

The study of leachate flow through a waste fill in saturated (ponding) conditions has been undertaken with different tracers commonly used in waste and soil studies. The use of multiple tracers has enabled individual characteristics of each tracer to be observed, but more importantly, allowed the trend of solute flow through waste to be better understood. A combination of gravimetric vertical flow and recycling flow conditions of tracers has enabled direct comparison of different flow regimes for a better understanding.

The time required for the modal mass flux of the tracers in the waste appeared to be less than 75% of the expected time for a modal mass flux for a uniform flow through the waste fill. There is however a similarity in the observed time for the median mass flux of the tracers and the calculated median time for a uniform mass flux from the waste, thereby suggesting that a portion of the solutes are relatively slower than the average flowrate in the waste fill. The mass recovery of the tracers indicates permanent staining (sorption) of the waste particles by the Coomassie Brilliant Blue, which in turn suggests the unsuitability of Coomassie Brilliant Blue for tracer tests in waste. Irreversible sorption of the particles of Lithium to the waste constituents or precipitation was also observed in the recycling flow test.

The recycling flow regime for tracer tests in waste has been found to be useful in determining the effective porosity of a waste fill. This implies increased accuracy in the formulation of waste models. Other benefits includes simplicity, shorter time and better efficiency in undertaking tracer tests in waste,

As in previous studies for unsaturated flow through waste, the non-uniformity in the solute flow through waste in submerged saturated conditions has been established and it constitutes probably of relatively fast advective flow particles and slow or temporarily immobile flow particles in "dynamic stores". It appears that the submerged saturation condition enhances solute storage and reversible sorption more than in unsaturated conditions. Unlike in previous studies, it has been demonstrated that solute particles flow through and may also be stored within waste constituents. Consequently, the formulation of flow models through waste in saturated conditions should incorporate secondary intra-particle flow of solute in addition to the usual matrix flow.

In general, the non-uniformity of waste should be considered when simulating the leachate produced from a waste fill or facility. Otherwise, misleading stabilisation period will be predicted, which may be may cause serious unrecoverable pollution, which is detriment to sustainable development of the environment

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