Research Journal of Applied Sciences, Engineering and Technology 2(4): 348-355, 2010
ISSN: 2040-7467

Submitted Date: March 25, 2010 Accepted Date: June 26, 2010 Published Date: July 10, 2010

Understanding Solute Transport in a Solid Waste Fill Using Basic Principles of Soil Mechanics and Stochastic Modelling

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Abstract: The solute transport owing to contaminant constituents in an emplaced solid waste was studied in saturated and oversaturated conditions using Sodium Chloride as the tracer. The flow was analysed using physical and probabilistic models commonly used in soil investigations. The Breakthrough Curves (BTCs) indicate relatively fast solute particles flowing vertically within the mass water flow in the waste fill. Unlike in previous studies, the transport of solutes from the inlet to the outlet has been undertaken using a transfer model, which is a function of volumetric water input. Although the simulated average fractional volume of water involved in the transport of solute particles appears to be slightly overestimated, the study indicates that virtually the entire water content of the saturated waste fill appears to be actively involved in solute transport within the waste mass. The disparity in the modelled and practicable values of the fractional flux in the saturated waste fill indicates that some of the solute particles are irreversibly adsorbed within the waste mass. The study further indicates solute flow along the cell wall in the oversaturated condition, and in general, shows the applicability of soil models to waste flows.

Key words: Inflow, leachate, saturation, solute transport, tracer test, transfer function

INTRODUCTION

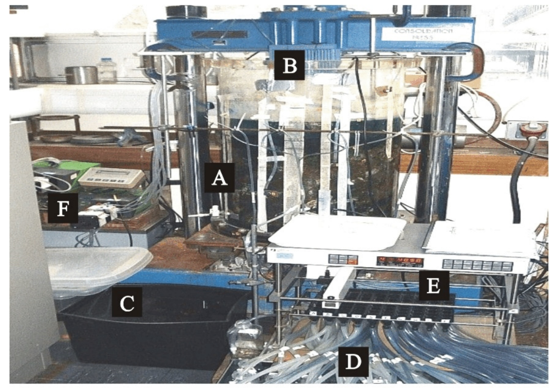
Approximately 15% of the world's population live in areas deficient of water. The majority of people in these areas struggle to obtain access to water to drink, keep clean, and meet their other needs to live. Two and a half billion people (more than a third of the world's population) have no access to improved sanitation, and more than 1.5 million children die each year from diarrheal disease (Fenwick, 2006). It is therefore imperative to reduce the contamination of the scarce water resources in these areas to avoid uncontrolled spread of water-borne diseases. Among the common sources of groundwater pollution are pollutants discharged into rivers or buried in landfill sites and deep repositories (Solomon and Powrie, 1994). The landfill of waste in both engineered landfills and refuse dumps is likely to continue for the unforeseen future. Although efforts have been geared towards drastic reduction of the recyclable and biodegradable wastes sent to landfill sites in the developed countries, the land still remains the only place where the remnants of processed wastes can be efficiently and economically kept in a safe manner on Earth (Ojha *et al.*, 2007). In the developing countries, indiscriminate dumping of waste is prevalent owing to poor environment awareness of the people and the non-commitment of the various governments to environmental protection. It is common for water-borne diseases that have since ceased in the West to still ravage various communities in these countries often resulting in epidemics (Ejechi and Ejechi, 2008). According to Fenwick (2006), the most prevalent seven water-related diseases in Africa are Hookworm, (198 million) Ascariasis (173 million) Schistosomiasis (166 million) Trichuriasis (162 million) Lymphatic filariasis (46 million) Onchocerciasis (18 million).

Efforts are often geared towards the minimisation or prevention of waste by increased recycling and compost of the biodegradable waste. In Europe, this has been undertaken using the Landfill Directive, enacted through the Council Directive 99/31/EC of 26 April 1999 on the landfill of waste, which entered into force on 16.07.1999 (Holgate, 2002). However, pollutants already exist in waste in old landfills, which were established prior to the stringent laws being enacted in the developed countries. Furthermore, rogue landfill and flash tipping still exist and may be difficult to be completely stopped. Once emplaced, the contaminants in waste are being leached by vertical flowing water forming "leachate" will may seep eventually into groundwater if not properly contained. Accurate delineation of a pollution plume is very difficult and recovery of a polluted subsurface resource is even more difficult and very expensive. Understanding the contaminant solute transport in a landfill will enhance leachate predictions, which in turn will enable landfill operators to manage leachate production through the provision of adequate leachate collection schedules and systems. The ability to estimate the volume of water actively involved in solute transport in the waste fill will enable strategies designed to quicken the biodegradation processes in the fill through effective recycling of water in the facility. In this way, effective utilization of the bio-gases produced into electricity is enhanced, changing the public perception of a landfill from a "nuisance" into a "noble" commercial investment (Sawyers, 1988).

Unlike soil whose characteristics have been studied and mastered using advanced theory and practical validations for a very long time, waste is relatively new and there have been not been universally accepted models for the behaviour of waste. This is not surprising as waste is heterogeneous in occurrence and in placement at a landfill site. In view of this, the majority of studies on waste have been undertaken using models initially formulated for soil. In many cases, reasonable results have been obtained as most of these studies focus on the macro-scale of the behaviour of waste. In case where discrepancies exist, logical reasons have been suggested, and have seemed reasonable (El-Fadel *et al.*, 1997; Oni and Richards, 2004).

In the majority of the studies undertaken on solute flux in waste fills, a preferential flow that resulted in non-uniformity of the vertical flow was generally reported (Rosqvist and Bendz, 1999; Beaven *et al.*, 2005; Oni, 2009). However, there have been different opinions concerning the quantification of the solute flux concentrations by these authors. Also, these have been different suggestions concerning the actual mechanics of the flow of the solute within the waste matrix. Some researchers believed that solute particles are carried though a mobile water fraction in macrospores and that immobile water exists in the other elements (parts) of waste and that water is only transferred from this element to the mobile element by diffusion (Beaven and Hudson, 2003). Furthermore, it was found out that by using Lagrangian modelling, that molecular diffusion should be ignored and that the non-uniformity in a waste fill could be best defined with a two-domain concept which assumes mobile water and thus solute advection in both the preferential and slow domain of the fill (Rosqvist and Destouni, 2000). Whatever the postulation, any investigation in which simulated and measured quantities appeared reasonable and in which results appeared pragmatic will further enhance the understanding of this complex material –waste.

The foremost significant research into solute transport in soil was undertaken more than two decades ago using a transfer function model (Jury, 1982). The reasonable validation of this postulated theory using field data (Jury and Stolzy, 1982) has made this model an important tool for simulating the transport of contaminant solute partlces and corresponding flow in waste studies (Rosqvist and Destouni, 2000; Rosqvist *et al.*, 2005). It is common to use conservative tracers for studies involving flow patterns in soil and waste (waste (Öman and Rosqvist, 1999; Pocachard, 2005). Although Lithium



A-Cell
B-ELAND framework
C-Water supply containter
D-Plastic tubing
E-Auto sampler
F-Peristaltic pumps

Fig. 1: Experimental set-up

appears to be the most common tracer used in waste, the use of fluorescent dyes and halides - chlorides, bromides and iodide are also prevalent owing to the ease of measurement and relative conservativeness in the waste. In this study, a stochastic function model, which utilises the water input as a variable has been used to estimate the active volumes of the water flux. Unlike previous studies, the transfer of solute from the waste surface through the waste fill to basal outlet is analysed as probability density function (pdf) of the cumulative water inflow. Furthermore, the tests were undertaken in both saturated and over-saturated conditions, in contrast to the unsaturated conditions used in earlier studies. Owing to its simplicity and reliability as a tracer, Sodium Chloride was used to simulate the contaminant solute in the waste fill throughout the experiments.

MATERIALS AND METHODS

The experiments described in this paper were undertaken by the author at the Waste Research Laboratory of the Department of Civil Engineering, University of Southampton, UK during the period 2005/2006.

The test cell and materials: The test cell (Fig. 1) comprised a cast acrylic cell, which is 0.480 m inner diameter, 0.90 m high, and has a wall thickness of 12 mm. The cell was mounted in an ELAN rig and has a circular galvanised iron mesh, placed on the upper surface of the emplaced waste in the cell. This prevented upward movement of lightweight particles during the upward flow of water in the cell. Manometer tubes were attached to three bores at the side of the cell at about 100, 250 and 400 mm above the cell base, respectively. The top platen on the waste was perforated with sixteen bores to serve as

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

inflow channels while eight similar bores were uniformly located at the base of the cell to serve as channels of outward movement of water from the sample.

The test material consists of a large-scale sample of a 15-year-old Municipal Solid Waste (MSW) materials obtained from the landfill site at Rainham, Essex, UK. The waste was cut to a maximum particle size of 40-50 cm and then mixed thoroughly according to the composition (Table 1) of the emplaced waste at the landfill. Initially, the top platen was lowered down on the waste fill to apply a load of 150 kPa until no rebound was noticed. During this time, the height of the waste fill decreased to 0.43 m. Following this, the top platen was placed 50 mm clear of the top of the cell to enable ease in monitoring the tracer in the surface pond overlying the waste fill in cell, in the oversaturated conditions.

Water recirculation, drainable porosity and saturated hydraulic conductivity: Water was pumped into the cell from a 50 L water container with two peristaltic pumps. The pumps had been calibrated to enable different inflow rates (L/h) to be easily and accurately obtained. The water supply distribution system comprised sixteen 3 mm diameter silicon tubes that were attached to the evenly placed bores on the top plate. The total length of these tubes from the pumps was approximately 1.7 m. Similarly, the water outflow from the cell passed through 3 mm diameter silicon tubes attached to the bores at the base of the cell. Each of these outlet tubes was 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 3 mm-diameter tube, 90 cm long, which is connected to a manifold that connects all outflow tubes to an auto sampling system, which includes 1 L glass containers. The top end of the Y connectors was open with a free water surface, 3.5 cm below the surface of the waste fill.

The waste was saturated sequentially from the bottom of the cell via the water distribution tubes. Water levels in the manometer tubes at each water increment were measured as soon as the levels stabilised. The waste was then drained and the same process was repeated with a downward direction of water flow. The porosity of the waste fill was calculated using the following equation

$$\text{Drainable porosity}_{\text{(waste)}} = (100/A)(dv/dh) \qquad (1)$$

where A is the surface area of the cell and dV/dh is the rate of volume change with respect to the hydraulic head in the waste fill.

Following re-saturation of the waste, the water inflow was increased steadily until the outflow rate was equal to the inflow rate of 1l/hr. At this period, the average hydraulic head was measured and used to determine the saturated hydraulic conductivity (k) using the Darcy's equation stated below:

$$k = Q / (A(\Delta H / L)) \qquad (2)$$

where Q is the volumetric flow rate through the waste sample and A is the cross sectional area of the waste sample. DH/L is the hydraulic gradient in the waste sample - DH is the change in hydraulic head and L is the length of flow.

Tracer tests: The tracer test was undertaken for saturated and oversaturation conditions. Prior to these, four samples of the effluent from the waste were collected in 25 mL bottles to establish the background concentration of tracer-Sodium Chloride in the waste fill. In the oversaturation condition, a surface pond whose volume enabled a downward gravitational flow of 1 L/h overlaid the waste in the cell. Following a steady state of top pulses of water flow in waste, the inlet tubes were quickly changed into the tracer container for continuous sprinkling of the tracer at the already established steady flow. The effluent from waste was collected by the auto-sampler using a 1 L glass container for each hour's collection. The tracer irrigation on the waste by pulses continued for 48 h prior to the washout of the waste by clean water. This continued for 363 h before the flow network was shut.

In the saturation condition, there was no surface pond and the water and tracer input into the waste was directly on the surface of the waste fill. Following a short period of shutdown in the previous tests, steady state conditions were established for a volumetric flowrate of 0.547 L/h, which has initially been determined as the discharge rate for full saturation of the waste fill. This continued for 48 h prior to the waste being flushed out by clean water. The entire tests were stopped following an elapsed time of 450 h - when background concentrations of the tracer appeared to have been reached. In both tests, the measured concentrations of the tracer were taken as that for the basal (exit) surface of the waste column.

Transfer function models - estimate of solute water flux: Ordinarily, a Transfer Function Model (TFM) is defined as a function of one variable (Jury, 1982; Box and Jenkins, 1976). Transfer function is often used in the analysis of single-input single-output systems and filters, especially in Control Engineering. The stochastic TFM has been shown to be related to the law of mass balance

Table 2: The physical properties of the waste fill

Waste properties	Dry density (kg/m^3)	Drainable porosity-downward (%)	Drainable porosity-upwards (%)	Effective field capacity (%)	Darcian flux (saturated) (m/s)	Darcian flux (oversaturated) (m/s)	Saturated hydraulic conductivity (m/s)
Values	722	7.5	8.8	37.7	8.40×10^{-7}	1.54×10^{-6}	1.65×10^{-6}

for solute transport in soil and waste (Jury, 1982; Rosqvist and Bendz, 1999). As such, parametric functions and non-parametric functions of physical properties of solute transport can be equated. Although the TFM was originally proposed for flow conditions in the vadose zone (unsaturated conditions), it has since been found to be applicable to saturation conditions in waste (reported in a paper in press). Under the prevailing conditions in the tests, the probability density function (pdf) of the travel time of the solutes is lognormal and can be represented for any measuring depth of the waste as (Jury *et al.*, 1986; White *et al.*, 1986; Rosqvist, 1999; Rosqvist and Bendz, 1999).

$$g(t) = [(2\pi^{0.5}\sigma t)]^{-1} e^{[-(In(t)-\mu)^2/2\sigma^2]} \qquad (3)$$

where t is the elapsed time since the start of flow; μ is the mean value of In(t); σ and σ^2 are the standard deviation and variance of In(t) respectively. With a constant inflow water pulse rate of i_0, it has been proposed that t in Eq. (1) can be replaced by the cumulative water inflow into the waste $I = i_0 t$ in Eq. (1) to give an equivalent pdf as below (White *et al.*, 1986).

$$f(I) = [(2\pi^{0.5}\sigma_I)]^{-1} e^{[-(In(I)-u_I)^2/2\sigma_I^2]} \qquad (4)$$

where, I is the cumulative water inflow into the waste; μ_I is the mean value of In(I); σ_I and σ_I^2 are the standard deviation and variance of In(I), respectively.

Equally, the hydro-physico (non-parametric) function based on mass balance that is equivalent to Eq. (3) and (4) can be represented as (White *et al.*, 1986).

$$f(I) = \frac{C(I)}{k} \qquad (5)$$

where C(I) is the exit concentration relating to I, and k is called the constant of normalisation that is equivalent to the mass of the solutes during the tests.
In general,

$$\int_0^\infty g(t)dt = 1 \quad ; \quad \int_0^\infty f(I)dI = 1 \qquad (6)$$

The median of the water inflow ($\bar{I}$) is expressed as:

$$\bar{I} = EXP(\mu_I) \qquad (7)$$

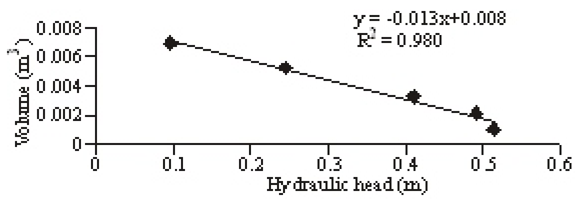


Fig. 2a: The volume against the hydraulic head of the water in waste fill - downward flow

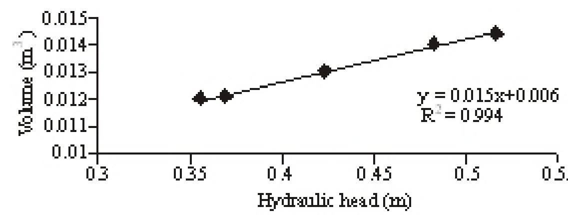


Fig. 2b: The volume against the hydraulic head of the water in waste fill - upward flow

The average fractional volume of water content in the waste participating in the solute transport can be estimated from the median of the water inflow as (White *et al*, 1986).

$$\theta_{st} = \bar{I}/AL = EXP(\mu_I)/(AL) \qquad (8)$$

where, L is the length or depth of measurement of the waste fill; A is the cross sectional area; is the volumetric content of the inflow usually expressed in litres or m^3.

RESULTS AND DISCUSSION

Throughout the tests, it was assumed that the solutes (tracer particles) and the flowing water have the same macroscopic velocity and the motion was in a downward vertical direction. The change in the hydraulic head of the water owing to change in volume of water in the waste fill for downward and upward direction is depicted in Fig. 2a, b, respectively. The gradient of these figures had been used for calculating the drainable porosity, using Eq. (1). The basic physical properties of the waste fill are summarised in Table 2. These values are typical of the hydro-physico properties of MSW landfills (Beaven, 2000; Oni, 2000). The field capacity is relatively high, indicating high content of absorbent materials. The small drainable porosity indicates highly compressible constituents in the waste fill.

The amount of water that passes through the waste can be expressed in terms of the Bed Volume (BV). The BV is the sum of the interstitial volume plus the intra particle pore volume in the waste. Under a steady and uniform flow, the solute flux in the waste fill is expected to pass as a plug flow vertically through waste and thus reach the basal waste surface when the input water is equal to a bed volume (Freeze and Cherry, 1979). Prior to this time, zero concentration of the solute (tracer) is expected from the outflow from the waste. However, the breakthrough curves for the waste (Fig. 3) did not depict a plug flow as significant quantities of the solute passed through the waste prior to the inflow of one BV in the fill. Equally, small quantities of solutes continued to flow from the exit following several BV quantities of inflow of "clean-water" used as washout of tracer from the waste. This indicates non-uniformity of the solute particles in the waste fill and perhaps indicate relative fast advection of some of the solutes and also relative slow movement of the other solutes. In addition to the flux, sorption and desorption of the solute particles to the waste may account for the observed long tail of the BTC. The modal fractional concentration has been attained at approximately 0.8 BV for the saturated flow and at approximately 1.7 BV for the oversaturated flow. This contrasts to a previous study in which the modal fractional concentration was attained at 0.11 and 0.04 BV for unsaturated waste emplaced in a pilot landfill and in a large-scale sample; with water contents of 0.27 and 0.43, respectively (Rosqvist and Destouni, (2000). It could be argued that the modal concentration in the unsaturated condition will be quickly achieved owing to a significant portion of the "advective" solute particles passing undiluted through the macropores. In the saturated conditions, the solute will pass through the matrix pores as well as the intra pores of the waste particles where dilution with inherent water in the absorbed waste constituents will reduce peak concentrations and increase the time and thus the amount of inflow equivalent of the bed volume required for the attainment of peak concentration. The increased BV required in the oversaturated is owing to the dilution of the tracer in the surface pond and waste-wall channel flows in addition to interparticle dilution earlier mentioned.

In an attempt to determine if the flow through the waste passes only through the drainable pores, the fractional concentration (measured concentration (C)/initial concentration (C_o) was plotted against the inflow equivalent of the bed volume that constitutes only the bulk drainable pores of the waste (BV^1) in Fig. 4. In this scenario, the peak concentration for saturated and oversaturated conditions is obtained at approximately 5 and 10 BV^1 respectively. In reality, one bed volume should normally be the maximum inflow equivalent volume prior to the attainment of peak concentration unless the bulk solute flux is severely retarded, which is not the case in the tests. The discrepancy from reality therefore suggests that the mass flux does not pass only through the drainable pores of the waste fill. In other words, it passes though the intra pore of the waste constituents as well.

The normalised concentration (per unit volume), calculated using Eq. (4) and (5), are plotted against the cumulative water inflow for the saturated flow in Fig. 5. The curve termed "model-pdf" is stochastic curve model fitting using the transfer function, described in Eq. (4) while the other curve is obtained using the observed solute measurement in the hydro-physico

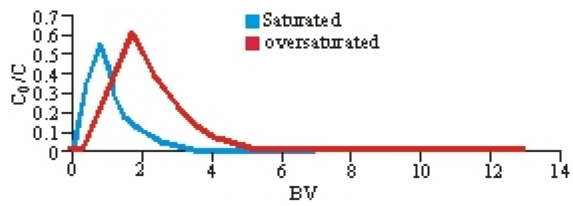


Fig. 3: The BTC in terms of the total bed volume of the waste fill

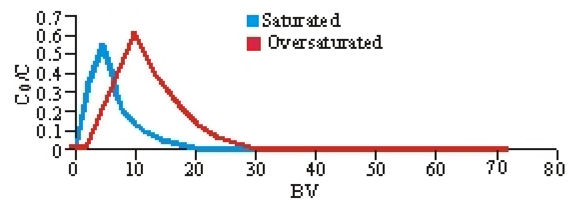


Fig. 4: BTC in terms of the drainable bed volume of the waste fill

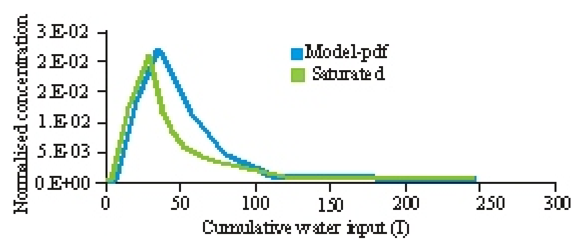


Fig. 5: The normalised flow against the cumulative water inflow - saturated conditions

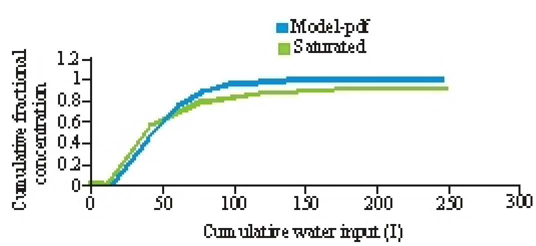


Fig. 6: The cumulative fractional flow against the cumulative water inflow - saturated conditions

relationship in Eq. (5). In general, these two curves appear to have a good matching, thus reiterating the equivalence of the parametric and non-parametric equation (Eq. 4 and 5). The cumulative fractional flow (Fig. 6) is calculated from Fig. 5 as a cumulative of the elemental fractional flow for each incremental water inflow using the trapezoidal rule.

The modelled and measured fractional flows for oversaturated conditions, calculated as for Fig. 5 and 6 are shown in Fig. 7 and 8. It appears that the curves of the modelled and measured normalised solute flux for the oversaturation conditions have a relatively good matching than in the saturated conditions. Accordingly, the modelled cumulative fractional flow for oversaturated condition fits reasonably well with measured values. Nevertheless, the degree of model fitting for both conditions appears reasonable enough for equivalence of the parametric and non- parametric (physical) functions, thereby enabling the quantification of the solute flux from stochastic parameters. It should be noted in the graphs depicted in Fig. 6 and 8 that the fractional flow/flux is quantitatively equivalent to the conventionally C/C_o used in waste and soil studies.

The pertinent data of the solute flux in the waste fill is summarised for both the saturated and oversaturated conditions in Table 3. The μ_I is obtained as described for Eq. (4). The median of the water inflow pdf ($\bar{I}$) is easily obtained from Eq. (7) and can also be obtained graphically from Fig. 6 and 8. The mean water content (dimensionless) of the waste that is involved in the solute transport (θ_{st}) has been calculated using Eq. (8). The average percentage of moisture content of the waste fill involved in the transport of solute particles is defined as the percentage of the average water content of the waste participating in the solute transport divided by the total water content in the waste. In a saturated condition, as in this study, the total water content is the total water content in the pores (i.e., the total porosity). The total waste porosity of the waste fill is 0.452. It can be seen that the percentage of moisture content participating in the flow of solute particles for the saturation condition using the modelled data exceeds 100% (Table 3), which is the maximum moisture content expected at full water saturation of waste fill. In a previous study (paper in press) using the time-travel time pdf described in Eq. (3), the water content of the saturated waste fill that is active in solute transport was found to be 97.57%.

Whereas, the use of cumulative inflow as the variable of the stochastic transfer function used to describe the solute transport in the waste fill has reasonably depict the characteristic trend of the solute, there appears to be a slight overestimation of the fractional volume of active water involved in the solute flux within the saturated waste fill. Some of the reasons may include: (a) the use of an input characteristic (inflow) to model an output characteristic (flux) that has undergone characteristic

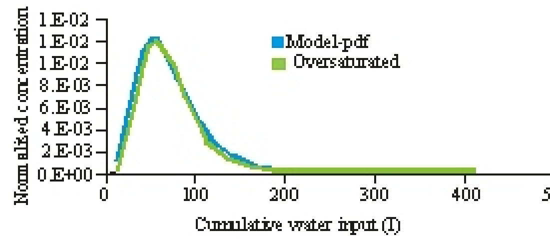


Fig. 7: The normalised flow against the cumulative water inflow - oversaturated conditions

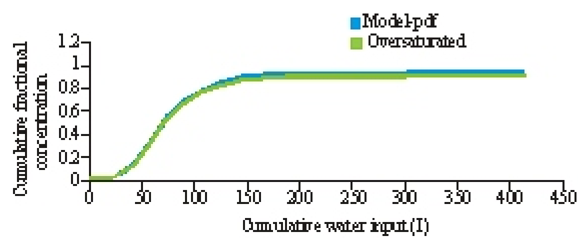


Fig. 8: The cumulative fractional flow against the cumulative water inflow - oversaturated conditions

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

Parameters/Quantities	Saturated	Oversaturated
μ_I	3.75	4.25
σ_I	0.25	0.548
$\bar{I}$ (m^3)	4.25E-02	7.01E-02
θ_{st} (m^3m^{-3})	0.546	0.901
θ_{st} (%)	120.9	199.3

changes (retardation) within the waste body; (b) probably, model fitting errors as visual in Fig. 6 and 8, and the additional parameter (i_0) introduced into the time parametric model (as in Eq. (4)) might increase the degree of error in the calculation the fractional water content of the waste fill that is participating in solute transport.

The percentage of water content participating in the solute flux in oversaturation conditions appears to be doubled what is expected for a normal saturation waste condition. As the surface pond remained constant throughout the Comparison of the volume of solute transporting water in saturated conditions in this study with reported values for unsaturated waste fills (Rosqvist and Destouni, 2000; Rosqvist *et al.*, 2005) shows that the solute transporting water increases with the degree of saturation of a waste fill. Similarly, comparison of the active water content involved in solute flux in waste fills with that in soils indicates that the solute transporting water increases with increasing intra-particle porosity of the porous media.

In general, techniques commonly used to estimate solute transport in soils has been found to be reasonably adequate for understanding and simulating solute flux in waste fills as well. Probably one of the reasons why these models appeared to give reasonable results in the modelling of hydro-physico properties of a waste fill is that both are generally porous material, although with variability in constituents. Although discrepancies exist in the inter-grain characteristics (porosity and hydraulic flow) of both media, these are only translated in terms of the relatively high solute water flow and field capacity of the waste fill.

CONCLUSION

The use of established hydro-physico models and stochastic models for soil has been found suitable for estimating solute flux in a waste fill. Unlike in previous studies, the transport of solutes from the inlet to the outlet has been undertaken using a transfer model, which is a function of volumetric water input.

The characteristic trend of the solute-water flux in the waste fill is reasonably depicted by the BTC using the cumulative water inflow in a stochastic transfer model. Intra-particle flux and non-uniformity in the flow of the solute in the waste was generally implied.

Generally, the fractional volume of water active in the transport of solute particles appears to increase with the degree of water saturation of the bulk waste fill. However, there appears to be a very slight overestimation of the predicted fractional volume of water content in the waste participating in the solute transport using the transfer model utilising the cumulative water as the variable the transfer function. The model has not been able to completely filter the characteristic changes undergone by the solute flux between the input and output lines of measurement. This could be eliminated by simulating with the time travel pdf, which utilises only the outflow parameter of the solute flux.

ACKNOWLEDGMENT

The author is profoundly grateful to Prof David Richards and Dr Richard Beaven, both of the Department of Civil Engineering, University of Southampton for the financial assistance and opportunity given to undertake these experiments ("Merci").

REFERENCES

Box, G.E.P. and G.M. Jenkins, 1976. Time Series Analysis. Holden-Day, San Francisco, USA.

Beaven, R.P. and A.P. Hudson, 2003. Description of a tracer test through waste and application of a dual porosity model. Ninth International Waste Management Symposium, Sardinia.

Beaven R.P., L. Dollar, O.A. Oni and N. Woodman, 2005. A Laboratory Scale Saturated and Unsaturated Tracer Test Through Waste. In: Gourc, J.B. (Ed.), International Workshop, Hydro-physico-mechanics of Landfills. Grenoble, France

Ejechi, E.O. and B.O. Ejechi, 2008. Safe Drinking water and satisfaction with environmental quality of life in some oil and gas industry impacted cities of Nigeria. Soc. Indic. Res., 85(2): 211-222.

El-Fadel, M., A.N. Findikakis and J.O. Leckie, 1997. Modeling leachate generation and transport in solid waste landfills. Environ. Technol., 18: 669-686.

Fenwick, A., 2006. Waterborne infectious diseases-could they be consigned to history? Science, 313(5790): 1077-1081.

Freeze, R. and J.A. Cherry, 1979. Groundwater. Prentice-Hall, New Jersey.

Holgate, G., 2002. Implementation of the EC landfill directive: The landfill (England and Wales) regulations 2002. Land Contaminat. Reclamat., 10(2): 101-105.

Jury, W.A., 1982. Simulation of solute transport using a transfer model. Water Resour. Res., 18: 363-368.

Jury, W.A. and L.H. Stolzy, 1982. A field test of the transfer function model for predicting solute transport. Water Resour. Res., 18(2): 1665-1675.

Jury, W.A., G. Sposito and R.E. White, 1986. A transfer function model of solute transport through soil 1, Fundamental concepts. Water Resour. Res., 22(2): 243-247.

Ojha, C.S.P., M.K. Goyal and S. Kumar, 2007. Applying Fuzzy logic and the point count system to select landfill sites. J. Environ. Monitor. Assess., 135(1-3): 99-106.

Öman, C. and H. Rosqvist, 1999. Transport of organic compounds with percolating water through landfills. Water Resour. Res., 33(10): 2247-2254.

Oni, O.A. and D.J. Richards, 2004. Estimating moisture volume in a municipal solid waste landfill during refuse infill. Proceedings of the 19th International conference on solid waste technology and management. Philadelphia, pp: 446-455.

Oni, O.A., 2000. An investigation into the impact of sequential filling on properties of emplaced waste lifts and moisture stored in a municipal solid waste landfill. Ph.D. Thesis, University of Southampton, Southampton.

Oni, O.A.G., 2009. Studying pollutant solute transport in saturated msw using multi-tracer tests. Aust. J. Basic Appl. Sci., 2(4): 3727-3740.

Beaven, R.P., 2000. The hydrogeological and geotechnical properties of household waste in relation to sustainable landfilling. Ph.D. Thesis, Queen Mary and Westfield College, University of London, London.

Pocachard, J., 2005. Application of Tracers Techniques to Landfill Observations. In: Gourc, J.B. (Ed.), International Workshop, Hydro-physico-mechanics of Landfills. Grenoble, France.

Rosqvist, H., 1999. Lutes transport through preferential flow paths in landfills. Proceedings Sardinia 99, Seventh International Landfill Symposium, Italy.

Rosqvist, H. and D. Bendz, 1999. An experimental evaluation of the solute transport volume in biodegraded municipal solid waste. Hydrol. Earth Syst. Sci., 3(3): 429-438.

Rosqvist, H. and G. Destouni, 2000. Solute transport through preferential pathways in municipal solid waste. J. Contam. Hydrol., 46 (1-2): 39-60.

Rosqvist, H., L.H. Dollar and A.B. Fourie, 2005. Preferential flow in municipal solid waste and implications for long-term leachate quality: valuation of laboratory-scale experiments. Waste Manage. Res., 23(4): 367-380.

Sawyers, R.D., 1988. The valuation of landfill sites. J. Prop. Valuat. Investment, 6(2): 119-126.

Solomon, C.L. and W. Powrie, 1994. Contamination of groundwater and the risk to human health. Int. J. Environ. Poll., 4(3/4): 283-293.

White, R., J. Dyson, R. Haigh, W. Jury and G. Sposito, 1986. A transfer function model of solute transport through soil: 2: Illustrative applications. Water Resour. Res., 22(2): 248-254.