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FRACTURED TILL: ITS IMPLICATIONS FOR CONTAMINANT IMPACT ASSESSMENT

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ABSTRACT

Some of the factors to be considered in performing impact assessments associated with proposals to locate landfill sites in fractured till are discussed. These factors include the effect of the mass of contaminant, infiltration into the landfill and the contaminant transport pathway on the contaminating lifespan of a landfill. The effects of fracturing of the till on the potential impact of contaminants on underlying aquifers are examined with respect to a hypothetical case. The influence of both a man-made (compacted clay liner) and natural intact clayey layer in contact with the fractured till is examined. The concept of developing "triggers" to initiate leachate control measures, and the associated potential impact on groundwater, is discussed in the context of the potential design life of the primary engineering (i.e. the underdrain system) in a landfill.

Keywords: environmental impact; contaminant migration; landfill; design; fractures, liners; groundwater; leachate collection

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INTRODUCTION

Unweathered clayey tills have been considered to represent a relatively ideal environment for the location of waste disposal sites. It is well recognized that "weathered" tills are fractured, however conventional hydrogeologic investigations of "unweathered" tills have typically implied that these tills are unfractured. Recent research and field investigations which have included angled boreholes or deep test pits, suggest that conventional investigations may be misleading and that many of these "unweathered" tills are indeed fractured to depths of as much as 10 m (e.g. Herzog and Morse, 1986; Ruland, 1988; D'Astous et al., 1989; Herzog et al., 1989; McKay, pers. comm.). Since these tills are frequently underlain by aquifers, this then raises the question as to what effect the fracturing will have on the potential impact of waste disposal sites, located in the fractured tills, upon underlying groundwater resources?

The modelling of the migration of contaminants in fractured porous media has received considerable attention (e.g. see Rowe & Booker, 1989a for a summary of recent research) and the concept of attenuation due to diffusion of contaminants from fractures into the matrix of the adjacent porous media is well established (e.g. see Freeze and Cherry, 1979). However, the migration of contaminant from landfill sites through fractured tills and into underlying aquifers has not received much attention.

Recently, Rowe and Booker (1989b, 1990) developed a semi-analytic technique for modelling contaminant migration from a landfill, through fractured media, and into an underlying aquifer. This model, which can be easily implemented and runs in a few seconds on a micro-computer, can be readily used to perform sensitivity studies associated with impact assessment. In parallel, Sudicky (1990) has developed a Laplace transform Galerkin technique which could be used for modelling migration through fractured media and into an underlying aquifer.

The objective of this present paper is to discuss some of the factors to be considered in performing impact assessments associated with proposals to locate landfill sites in fractured till. Factors to be considered include the effect of the mass of contaminant, infiltration into a landfill, and the contaminant pathway on the "contaminating lifespan" of a landfill. The potential effect of fractures on the time of arrival of contaminants in an underlying aquifer and the influence of both a man-made (compacted) and natural, intact clayey layer in contact with the fractured till will be illustrated with reference to a hypothetical landfill example. Attention will be focussed on landfills for municipal and non-hazardous industrial waste.

It is emphasized that this paper is concerned with the potential for migration of contaminants which occur in solution at relatively low concentrations (e.g. as in domestic waste leachate). The analysis of the migration of contaminants, through fractured media, from hazardous industrial waste facilities, and in particular, the migration of concentrated dense non-aqueous phase chemicals requires consideration of additional factors and is beyond the scope of this particular paper.

CONTAMINATING LIFESPAN AND FINITE MASS OF CONTAMINANT

The "contaminating lifespan" of a landfill may be defined "as the period of time during which the landfill will produce contaminants at levels that could have unacceptable impact if they were discharged into the surrounding environment" (MOE, 1988). When dealing with groundwater contamination, it is necessary to consider the transport pathway (and consequent attenuation) when assessing the contaminating lifespan. This will, of course, vary from one landfill to another.

The contaminating lifespan of a landfill will depend, inter alia, on the mass of contaminant per unit area (i.e. the height of landfill), the infiltration and the pathway for contaminant release. The higher the landfill, the greater the mass of any given contaminant and, all other things being equal, the longer the contaminating lifespan. The greater the infiltration (and hence volume of leachate collected) the shorter will be the contaminating lifespan since there is greater opportunity for contaminant to be leached out and treated. The greater the potential for attenuation along the escape pathway, the shorter the contaminating lifespan.

The simplest case is that of a conservative contaminant species which is highly soluble, readily leached from the waste and does not decay due to biological activity in the landfill.

For waste disposal sites such as municipal landfills, the mass of any potential contaminant within the landfill is finite. The process of collecting and treating leachate involves the removal of mass from the landfill and hence a decrease in the amount of contaminant which is available for transport into the general groundwater system. Similarly, the migration of contaminant through the underlying deposit also results in a decrease in the mass available within the landfill. For a situation where leachate is continually being generated (e.g. due to infiltration through the landfill cover), the removal of mass by either leachate collection and/or contaminant migration will result in a decrease in leachate strength with time.

The peak concentration, c_o, of a given contaminant species can usually be estimated from past experience with similar landfills. The total mass of contaminant is more difficult to determine. Nevertheless, upper bound estimates can be made by considering the observed variation in concentration with time at landfills where leachate concentration has been monitored or by considering the composition of the waste.

Until fairly recently, there has been a paucity of data concerning the available mass of contaminants within landfills; however, this situation is changing now that many landfills have leachate collection systems. Given that concentration is simply mass per unit volume, the mass of a given contaminant collected in a given period is equal to the concentration multiplied by the volume of leachate collected. By monitoring how this mass varies with time, it is then possible to estimate the total mass of that species of contaminant within the landfill. In the absence of this information, studies of the composition of waste (e.g. Cheremisinoff & Morresi, 1976; Kirk & Law, 1985; Hughes et al., 1971) can be used to estimate the mass of given contaminant or groups of contaminants. For contaminant species predominantly formed from breakdown or synthesis of other species (e.g. by biological action), an upper bound estimate of the mass of contaminant may be obtained from the estimates of the mass of chemicals which go to form the derived contaminant.

For the purposes of modelling the decrease in concentration in the leachate due to movement of contaminant into the collection system and through the barrier, it is convenient to represent the mass of a particular contaminant species in terms of a "reference height of leachate", H_r, as described below.

If the initial mass, m_{TC}, of a contaminant species (e.g. chloride) can be estimated, then the reference volume of leachate which would contain this mass at an initial concentration c_o is

$$[1] \qquad V_{TC} = \frac{m_{TC}}{c_o}$$

In general, this volume will not correspond to the actual volume of leachate because it is based on the conservative assumption that all this available mass can be quickly leached from the solid waste. It is convenient for both mathematical and physical reasons to express the volume V_{TC} in terms of a reference height of leachate "H_r" which is defined as the reference volume of leachate divided by the area, A_o, through which contaminant passes into the primary "barrier"

$$[2] \qquad H_r = \frac{V_{TC}}{A_o}$$

or

$$[3] \qquad H_r = \frac{m_{TC}}{c_o A_o}$$

where H_r = representative height of leachate [L]

m_{tc} = total mass of a contaminant species of interest [M]

c_o = peak concentration of that species in the landfill [ML^{-3}]

A_o = area through which contaminant can migrate into the underlying layer [L^2]

and H_r represents the mass of contaminant available for transport into the soil and/or collection by the leachate collection system.

For the case where there is an infiltration q_o through the landfill cover and where all the leachate is collected (i.e. no migration into the underlying soils), it can be mathematically shown that the process of dilution within the landfill gives rise to a decrease in the average leachate strength with time (recognizing that leachate concentrations may vary seasonally) such that the concentration in the landfill at time t, $c_{oL}(t)$ is related to the peak concentration c_o by the relationship

$$[4a] \qquad c_{oL}(t) = c_o \exp\left(\frac{-q_o t}{H_r}\right)$$

or, on rearranging terms, the time required for the leachate strength to reduce to some specified value, c_{oL}, is given by

$$[4b] \qquad t = \frac{-H_r}{q_o} \ln\left(\frac{c_{oL}}{c_o}\right)$$

where H_r is the reference height of leachate [L] and q_o the infiltration [LT^{-1}] as previously defined.

To illustrate the implications of Eq. (4a), consider a landfill where the average thickness of waste, H_w = 10 m, the average dry density of that waste is ρ_{dw} = 500 kg/m^3 and where chloride is assumed to represent 0.2% of the dry weight of the waste (i.e. p = 0.2%), then the mass of chloride per unit area, (m_{tc}/A_o), is given by $m_{TC}/A_o = H_w \rho_{dw} p$ = 10x500x0.02 = 10 kg/m^2. If the peak concentration of chloride is 1000 mg/L (1 kg/m^3) then the "reference height of leachate", H_r, is given by

$$H_r = \frac{m_{TC}}{A_o c_o} = \frac{10}{1} = 10 \text{ m}$$

Assuming an infiltration through the cover of 0.15 m/a, the decrease in chloride concentration with time, simply due to dilution in the landfill, can be calculated from Eq. (4a) as indicated for Case [a] in Figure 1. For this particular example, the chloride level would reduce to 250 mg/L after approximately 90 years and (from Eq. 4a) to 125 mg/L after about 140 years.

It is noted that if the infiltration was 0.3 m/a (all other factors being equal), the concentration would decay much faster (see Case [1b], Figure 1) and would reduce from the peak value of 1000 mg/L to 250 mg/L after about 45 years and to 125 mg/L after about 70 years.

If chloride represents 0.1% of the waste (e.g. see Hughes et al., 1971), rather than the 0.2% assumed above, then m_{Ic} = 0.001x10x500 = 5 kg/m^2 and H_r = 5 m. For an infiltration of 0.15 m/a, this gives Case [c] for which the results shown in Figure 1 are precisely the same as those obtained for case [b] (since the ratio H_r/q_o is the same). Clearly, the same result would also be obtained for 5 m of waste if chloride represented 0.2% of the waste (i.e. m_{Ic}/A_o = 5x500x0.002 = 5 kg/m^2).

If one were to assume the same total mass of chloride m_{Ic} as in case [a] above, but if the peak concentration c_o = 2000 mg/L (2 kg/m^3) then

$$H_r = \frac{m_{IC}}{A_o c_o} = \frac{10}{2} = 5 \text{ m}$$

and the decrease in concentration with time given by Eq. (4a) for q_o = 0.15 m/a is as shown by Case [d] in Figure 1. In this case, the concentration decreases from the peak value of 2000 mg/L to 250 mg/L in about 70 years and to 125 mg/L in just over 90 years.

Equation (4a) only considers decreases in concentration due to dilution. Clearly, if the contaminant experiences other decay mechanisms (e.g. biological decay), then the rate of decrease with time may be faster than implied by Eq. (4a). Similarly, if contaminant is lost to the underlying stratum (e.g. due to diffusion), the rates of decay may be faster than implied by Eq. (4a).

When considering the contaminating lifespan of a landfill, it is necessary to define what is meant by "unacceptable impact". In the Province of Ontario, the Ministry of the Environment has a "Reasonable Use" policy (MOE, 1986) and if a "reasonable use" for groundwater was as drinking water then an "unacceptable impact" could be interpreted as an increase in chloride (say) which exceeds that defined by the "Reasonable Use" policy. This would mean an increase in chloride level at the site boundaries of 125 mg/L (or less if there are background levels of chloride in the groundwater).

If one adopts this definition of "unacceptable impact", then for the examples considered in Figure 1 it would be necessary for the leachate collection system to operate for between a maximum of about 140 years for Case [a] and a minimum of about 70 years (for Cases [b] and [c]) if dilution of leachate was the only available attenuation mechanism. In reality, of course, contaminants will generally have to pass through some form of

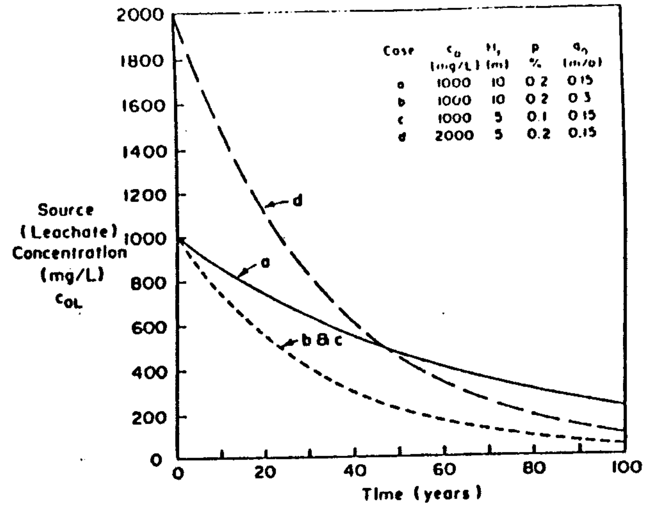

FIGURE 1 DECREASE IN LEACHATE (CHLORIDE) CONCENTRATION WITH TIME

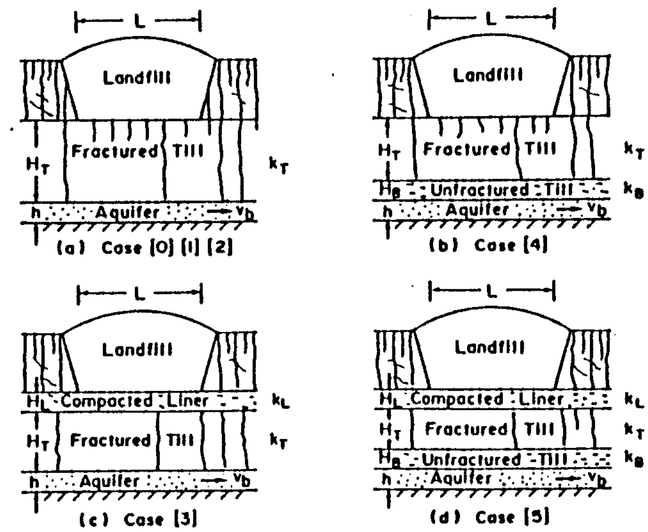

FIGURE 2 CASES CONSIDERED

hydrogeologic "barrier" (e.g. a clayey till) before entering the aquifer. The question then arises as to how much attenuation may occur as it passes through this "barrier" and into any underlying aquifer. Thus when considering contaminant impact on an underlying aquifer, the contaminating lifespan depends not only on the decay within the leachate but also on the potential attenuation in the soils between the landfill and the aquifer. This in turn will depend on the geometry of the landfill, the base elevation of the landfill, the head difference between the leachate and underlying aquifer, the properties of the till and the properties of the underlying aquifer. Of these, the most important are the hydraulic conductivity of the till and the head difference between the landfill and aquifer. The implications of fracturing of till and engineering remedies (e.g. compacted clay liners) will be discussed in the following sections.

HYDROGEOLOGIC SETTING AND HYDROGEOTECHNICAL DESIGN

The basic situations to be considered are shown, in cross-section, in Figure 2. Typically, two hydrogeologic environments may be encountered. Figure 2(a) shows a landfill separated from an aquifer by a thickness H_f of fractured till which has a bulk hydraulic conductivity k_f. In this case the fracture frequency decreases with depth but some fractures extend through the entire thickness of the clayey till. Figure 2(b) shows a similar situation except that in this case the fracture frequency decreases until at some depth, H_f, below the base of the landfill the fractures terminate. Thus the fractured till is underlain by an unfractured clayey till or clay layer, of thickness H_s and hydraulic conductivity k_s, which in turn overlies the aquifer.

These basic hydrogeologic environments may be enhanced by the design and construction of a suitable compacted clay liner, of thickness H_L and hydraulic conductivity k_L, as shown in Figures 2(c) and 2(d).

For the purpose of the following discussion, it is assumed that the fractured till, the compacted liner, the unfractured till and the aquifer shown in Figure 2 have the properties defined in Table 1.

In the natural setting (i.e. before construction of a landfill), there could be either upward or downward gradients from the aquifer to the groundwater table. In any event, the construction of a landfill can substantially change the flow regime in the vicinity of the landfill. In many cases, the elevation of the base of the landfill and the layout of the leachate underdrain system can be designed to provide gradients into the landfill (e.g. from the underlying aquifer) creating a "hydraulic trap" which will restrict the outward migration of contaminants to outward diffusion which can occur in opposition to the inward velocity (e.g. see Rowe, 1988) and migration will be primarily through the matrix of the till.

If there is a hydraulic trap then, under the most adverse conditions, the advective flow into the landfill will be entirely through the fractures and so migration through the matrix would be by pure diffusion. Clearly, this can be modelled as diffusive transport through this matrix, without the need to model fractures. To illustrate this, consider contaminant migration from a landfill where the characteristics of the leachate and collection system are as discussed in the previous section for Case [a] (i.e. $H_r = 10$ m; $c_o = 1000$ mg/L; $q_o = 0.15$ m/a). It is assumed that the base of the landfill is separated from a 1 m thick underlying aquifer by 4 m of fractured till (i.e. $H_t = 4$ m, as per Figure 2a; see Table 1, Cases [0], [a] for a full set of parameters). The concentrations of chloride in the aquifer (assuming negligible flow in the aquifer, i.e. $v_b = 0$) due to pure diffusion from the landfill can be calculated (allowing for finite mass of contaminant and the leachate collection system) using simple computer programs such as POLLUTE v5 (Rowe & Booker, 1983-90). From the results for Case [0a], shown in Figure 3, it is evident that with a working leachate collection system the first arrival of chloride in the aquifer at 1% of the initial leachate value occurs after about 70 years. The concentration then increases until a peak impact of just under 100 mg/L is reached after about 275 years. Assuming that the initial background concentration of chloride in the aquifer of less than 50 mg/L, this increase of 100 mg/L would meet Ontario's "Reasonable Use" Guidelines (MOE, 1986). However, this analysis assumes that the leachate collection system operates for about 140 years (i.e. until the concentration in the leachate, c_{ot}, is less than the maximum increase in chloride permitted in the aquifer; this time can be calculated using Eq. (4b) viz. if $c_{ot}/c_o = 0.125$, $H_r = 10$ m; $q_o = 0.15$ m/a, then $t = -H_r/q_o \ln(c/c_o) = 139$ years), and hence implies a contaminating lifespan of about 140 years.

The question which then arises is as to whether the leachate underdrain system will function for 140 years and what would be the impact of a failure of the leachate collection system prior to this time. The following sections will focus on this question and the implications that fracturing of the till might have on the determination of an answer.

Leachate Monitoring and "Trigger" Levels

Contingency measures for landfills have typically been intended to remove contaminant (e.g. by installing purge wells in an aquifer) if it "unpredictably" makes its way through to the aquifer. Thus in the example considered in the previous section, purge wells in the aquifer would represent a contingency. If the assumed parameters are valid and the leachate underdrain system lasts more than 140 years then this contingency would not be needed.

In this author's opinion, the assumption that a leachate underdrain system could function well enough to maintain a hydraulic trap for 140 years is questionable given the paucity of long term data on these systems. On the contrary, it would seem reasonable to anticipate that a leachate underdrain system could not be expected to function adequately for 140 years and if one accepts this hypothesis then a consequent buildup of leachate and potential contamination of the aquifer would not be regarded as "unpredictable". Under these circumstances, it would be appropriate to anticipate that a failure of the underdrain system would occur and to design an alternative means of controlling leachate levels once such a failure did occur (rather than relying on purge wells in an underlying aquifer to collect contaminants after they reached the aquifer).

Accepting that, at some time, failure of a leachate underdrain system will occur, it is necessary to monitor leachate levels to detect this failure and to have a "trigger" in terms of leachate level and concentration at which alternative leachate control (e.g. leachate wells installed in the waste) would be initiated. The "trigger" levels of leachate mounding will vary from one landfill to another and will, inter alia, depend on the level of attenuation which can occur between the base of the landfill and the underlying aquifer. This in turn will depend on the engineering (e.g. the presence of a compacted clay liner) and the hydrogeotechnical characteristics of the underlying strata (e.g. the level of fracturing, the hydraulic conductivity of the different strata, etc.). Based on the available field data, calculations can be performed to consider these factors to establish "triggers" in terms of leachate levels and concentrations at which leachate control measures would be initiated. These trigger levels may be based on the requirement that the consequent initiation of leachate control measures would prevent predictable but unacceptable impact on any underlying groundwater resource. This will be illustrated in the following sections.

Failure of a Leachate Underdrain System: 4 m Thick Fractured Layer

For the purpose of the following example, suppose that a landfill is being designed to operate with a hydraulic trap and that under these circumstances the impact is acceptable (e.g. for Case [0a] discussed in an earlier section). Suppose that the hydraulic trap operates for a period of time, t_o, and then fails and a downward gradient develops between the increased leachate level in the landfill and the underlying aquifer. The level of mounding in the landfill will depend on the dimensions of the landfill, the location of perimeter drains and the location of contingency leachate wells which may be installed in the landfill to control the level of mounding. The failure of the leachate collection system will take a period of time and the consequent gradual change in advective velocity can be modelled (e.g. using program POLLUTE v5), however for simplicity of presentation in this paper, the change in velocity is considered to occur quickly at time t_o which approximately corresponds to the mean time between when the failure began to occur and when the full downward gradient was developed.

Once downward gradients develop, the hydraulic conductivity and fracturing of the underlying till become critical. Suppose, for the sake of discussion, that the hydraulic conductivity of the fractured till had been determined from a pump test to be 10^{-7} cm/s and that, due to the nature of the field test, this incorporates the effect of any fractures on the bulk hydraulic conductivity. The Darcy velocity is the product of the bulk hydraulic conductivity and the gradient. For determining the gradient, it is initially assumed that the failed leachate level corresponds to a difference in head, Δh, of 6.35 m between the base of the landfill and the aquifer (Case [1]).

Given the bulk hydraulic conductivity of the fractured unweathered till, there are two possible bounding situations with regard to the effect of the fractures. On the one hand, the fractures may not be significant conduits for contaminant transport and contaminant migration may simply occur through the matrix of the till; this will be modelled as a conventional porous medium, without explicitly considering the fractures, using the theory proposed by Rowe and Booker (1985; 1987) and program POLLUTE v5 (op. cit.).

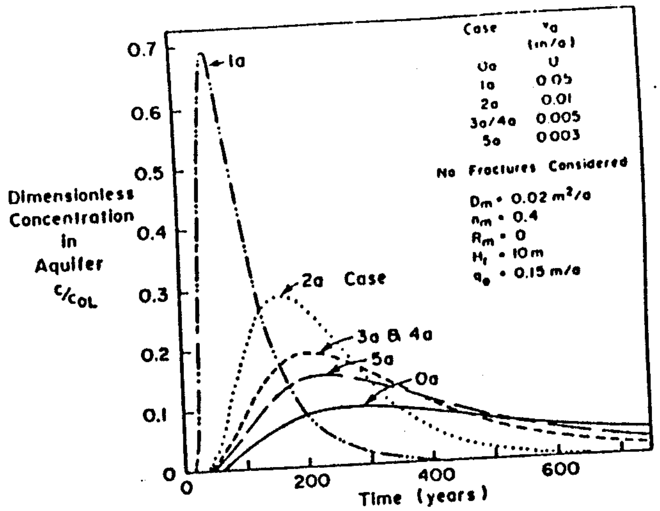

FIGURE 3 VARIATION IN CHLORIDE CONCENTRATION IN THE AQUIFER ASSUMING MIGRATION THROUGH THE MATRIX OF THE TILL (NEGLECTING FRACTURES)

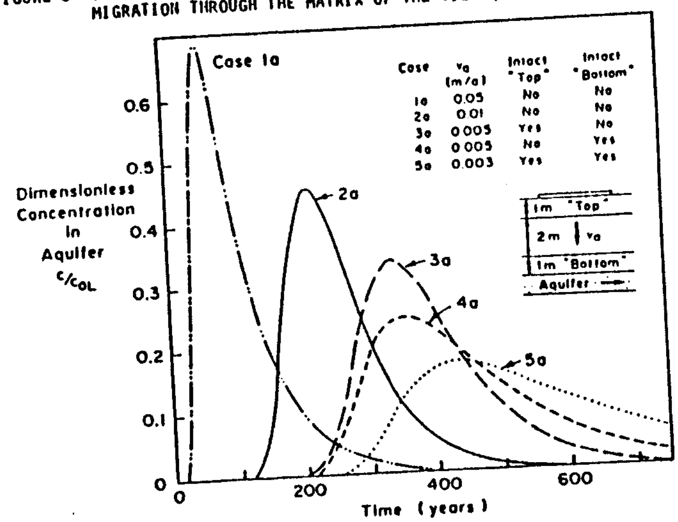

FIGURE 4 VARIATION IN CHLORIDE CONCENTRATION IN THE AQUIFER ASSUMING MIGRATION ALONG FRACTURES

On the other hand, the fractures may control migration and it can be assumed that all migration to the aquifer occurs through the fractures and none through the matrix (although attenuation may still occur due to matrix diffusion from the fractures into the adjacent clayey till). This will be modelled using the theory proposed by Rowe and Booker (1989; 1990) and program POLLUTE v5. Many situations will lie between these bounding cases, but by modelling these cases it is possible to obtain a reasonable engineering estimate of potential impact.

Figure 3 shows the results obtained assuming contaminant migration through the matrix for the five failure cases being considered, using the theory proposed by Rowe and Booker (1985; 1987). Figure 4 shows the corresponding results considering contaminant migration through fractured media and were obtained using the theory proposed by Rowe and Booker (1989; 1990). For the failure cases, the time shown on the Figures (e.g. Figs. 3 and 4) represents the time after failure (i.e. after time t_o as defined earlier).

Case [1a] corresponds to a major failure of the leachate collection system and a landfill underlain by 4 m of fractured material (see Figure 2(a) and Table 1. Assuming migration through the matrix of the till, Figure 3 shows first arrival of contaminant in the aquifer (at 0.01% of the source value at the time the failure occurred) within 20 years following the failure. The concentration in the aquifer increases with time reaching a peak value of about 69% of the value in the leachate at the time of failure, approximately 52 years after the failure occurred. Assuming migration along the fractures (with diffusion into the adjacent matrix), Figure 4 also shows first arrival of contaminant in the aquifer within 20 years of the failure, rapidly increasing to a peak value of about 69% of the leachate concentration at failure after 54 years. In both cases, the concentration decreases after the peak value has been reached.

Comparison of the results for Case [1a] in Figures 3 and 4 shows very similar results irrespective of whether it is assumed that migration is through the matrix or through the fractures and the attenuation factor is 0.69 (i.e. the peak concentration in the aquifer is 0.69 times the value in the leachate at the time of failure). The spacing of fractures does have some influence on the results, however it is clear that irrespective of which analysis is used there will be a substantial impact about 50 years after failure.

If it is assumed that the maximum allowable increase in concentration in the aquifer is 125 mg/L for chloride, then the results for Case [1a] can be used in conjunction with Eq. (4b) (or Figure 1) to estimate how long the leachate collection system would have to work before a failure and mounding to the level implied by Case [1a] could be allowed to occur. For example, taking the allowable increase in concentration to be 125 mg/L and an attenuation factor of 0.69 (based on Figures 3 and 4), the allowable value in the leachate at the time of failure would be $c_{ot} = 125/0.69 = 181$ mg/L. This result (and similar results for other levels of mounding) can be used to construct an envelope of trigger levels at which control measure would be required. Thus Case [1a] corresponds to a differential head $\Delta h = 6.35$ m and a maximum allowable concentration in the leachate of 181 mg/L. This is plotted on Figure 5 (which will be discussed subsequently). Assuming

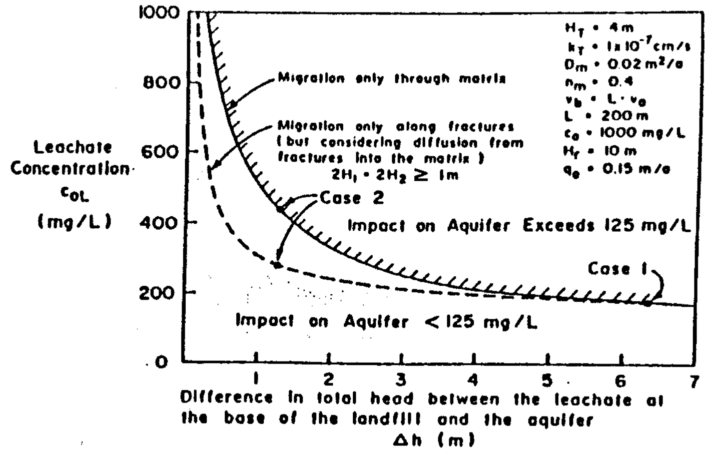

FIGURE 5 "TRIGGER" CONCENTRATION LEVELS FOR LEACHATE PUMPING AT VARIOUS LEVELS OF MOUNDING FOR THE HYPOTHETICAL CASE BEING CONSIDERED landfill conditions [a] (c_o = 1000 mg/L, H_r = 10 m, q_o = 0.15 m/a), the time at which this failure could occur can be estimated from Eq. (4b), viz.

$$t = \frac{-H_r}{q_o} \ell n\left(\frac{c_{oL}(t)}{c_o}\right) = \frac{-10}{0.15} \ell n\left(\frac{181}{1000}\right) = 114 \text{ years}$$

If the failure occurs prior to this time then the eventual impact on the aquifer would be unacceptable based on the assumed conditions in this example.

It is probably unrealistic to expect the leachate underdrain system to be fully functioning for 114 years and so these results indicate that some measures would have to be taken to decrease impact on the underlying aquifer if a failure of the underdrain system were to occur in less than 114 years.

One option would be to install leachate wells and lower the level of leachate mounding. To illustrate the potential effect for the fractured system shown in Figure 2a, it is assumed in Case [2a] that the difference in head between the leachate level at the base of the landfill and the value in the aquifer is 1.27 m (i.e. one fifth of that assumed in Case [1a]); all other parameters are identical. Figures 3 and 4 show the results assuming that the contaminant moves through the matrix and the fractures respectively. In this case, the assumption made concerning the mechanism of transport has a greater effect on the results than was the situation for Case [1a]. Referring to Figure 3, it is seen that for migration purely through the matrix, contaminant reaches this aquifer at the 1% level within 45 years and the concentration increases to a peak value of about 0.29 c_{oL} of the leachate concentration at failure about 160 years after failure of the leachate collection system.

Referring to Figure 4, it is seen that the modelling migration through the fractures (but considering attenuation due to matrix diffusion) gives a much later first arrival in the aquifer at the 1% level (i.e. after about 125 years) but in this case the peak impact is 0.45 c_{oL} about 215 years after failure. If the fracture spacing is not less than the 1 m value assumed in this calculation, then the lower and upper bound on the peak impact for the combination of parameters associated with Case [2a] are 0.29 (Figure 3) and 0.45 (Figure 4) times the value in the leachate at failure. Thus, in order for the impact on the aquifer not to exceed 125 mg/L, the concentration in the leachate at failure would have to be less than 277 mg/L based on Figure 4, or less than 430 mg/L based on Figure 3. These are plotted for Δh = 1.27 m in Figure 5. Using Eq. (4b) or Figure 1, it can be shown that to meet these requirements, the leachate collection system must work for between 56 and 85 years.

If it is considered to be unreasonable to expect that the leachate collection underdrain will maintain the hydraulic trap this long then either the level of leachate mounding would have to be further reduced (thereby decreasing the head difference, Δh, and hence the downward gradient and Darcy velocity) or some other engineering would be required. By repeating the calculations discussed above for different assumed levels of leachate mounding (and considering the corresponding difference in head between the waste and the aquifer), an envelope of trigger conditions can be constructed as shown in Figure 5. By monitoring the leachate levels and concentrations and comparing with the results shown in Figure 5, it would be possible to determine whether supplementary leachate control (e.g. leachate wells) would be required for the case being considered here. If the combination of leachate mounding and concentration plot below the dotted line then the impact on the aquifer is expected to be less than 125 mg/L and to be acceptable (for this case). If the combination of mounding and leachate concentration plots above the full line then future unacceptable impact may be anticipated unless some leachate control measures are taken. The zone between the dashed and full lines represents the range of variability associated with the extent to which contaminant migrates through the fractures and through the matrix of the fractured till. It would be conservative to use the lower curve as the trigger for leachate control measures.

Effect of a Compacted Clay Liner Over 3 m of Fractured Till

In the previous section, consideration was given to 4 m of fractured till with hydraulic conductivity of 1×10^{-7} cm/s. The attenuation characteristics of this deposit can be improved by removing the top 1 m of the fractured till and replacing it with a compacted clay liner having a hydraulic conductivity of 2×10^{-8} cm/s. Provided that the hydrogeotechnical studies show that it is suitable, the liner might be constructed by recompacting the till (at an appropriate water content). This corresponds to the situation shown in Figure 2. Assuming the same level of leachate mounding as in Case [2a], the results for Case [3a] shown in Figures 3 and 4 indicate that the clayey liner increases the time to first arrival in the aquifer, the time to peak impact and the magnitude of the peak impact. The peak impact evident from Figures 3 and 4 was 0.19 c_{oL} and 0.33 c_{oL} respectively.

The results presented in Figure 4 considered diffusion through the matrix of the 1 m thick liner and then transport along the fractures (with diffusion from the fractures into the matrix) in the lower 3 m of the deposit. This analysis assumes that once the contaminant breaks through the liner it can move to the fractures; this is equivalent to assuming that there is a thin permeable layer between the liner and the fractured till (e.g. a thin sand layer). If this permeable zone does not exist then the migration can be expected to be much closer to those given in Figure 3 than those in Figure 4 for this case.

Based on the results presented in Figures 3 and 4 for Case [3a] and Eq. (4b) (or Figure 1) in order to keep the impact on the underlying aquifer to less than 125 mg/L, it would be necessary for the leachate underdrain system to function between 30 and 65 years, provided that when the failure did occur leachate pumping was initiated to ensure that the difference in head between the landfill and the aquifer did not exceed the 1.27 m value assumed in the analysis.

Failure of a Leachate Underdrain System: 3 m Thick Fractured Layer

It appears that in many practical situations the unweathered fractured till may be underlain by an unfractured till or clay layer as indicated in Figure 2(b). To illustrate the effect of this layer, Case [4a] is examined in which it is assumed that this lower layer is 1 m thick with a hydraulic conductivity of $2x10^{-8}$ cm/s. It happens that the Darcy flow through this system is identical to that for Case 3 where a 1 m thick compacted clay liner (with $k = 2x10^{-8}$ cm/s) was considered. As a consequence, the results obtained assuming migration through the matrix of the fracture till (as shown in Figure 3) are identical to those obtained for Case [3a]. When migration through the fractures is considered, there is a difference between the results for Case [3a] and [4a]. It is evident from Figure 4 that it is more beneficial to have a 1 m thick natural intact layer at the bottom of the fractured till than it is to install a hydraulically similar compacted liner on top of the fractured till; this finding should not be extrapolated to other situations without direct verification. It is assumed in Case [4a] that when contaminant reaches the bottom of the fractures it will spread out and migrate evenly through the intact layer (as in Case [3a], this is equivalent to assuming that there is a thin permeable layer between the fractured till and the unfractured till). Based on the results given in Figures 3 and 4, the peak impact for Case [4a] lies between 0.19 c_{ot} and 0.24 c_{ot} and, based on Eq. (4b) or Figure 1, the length of time that the leachate collection system must operate in order to keep the impact on the aquifer to less than 125 mg/L is between 30 and 44 years.

Assuming that there is a 1 m thick intact layer beneath the fractured layer, it is of some interest to see what would be the effect of removing and recompacting 1 m of fractured till beneath the base of the landfill as shown in Figure 2(d). Analyses were performed assuming the parameters given for Case [5a] as given in Table 1. The results are shown in Figures 3 and 4. Assuming that migration is only through the matrix (i.e. that the contaminant can not spread out and move down through the fractures in the layer between the liner and the unfractured till), it is found that the peak impact is about 0.15 c_{ot} at about 225 years after the failure of the underdrain system. If one considers migration only through the fractures in the fractured layer (as per Figure 4), then the peak impact is slightly higher at 0.17 c_{ot} about 430 years after failure of the underdrain. Although the magnitude of impact is very similar, the earlier impact time (Figure 3) is likely to be more realistic for the situation shown in Figure 2(d). Based on these results, the leachate underdrain system would have to operate for between 12 and 20 years before failure in order to keep the increase in chloride in the aquifer to below 125 mg/L (assuming that after failure leachate pumping was initiated to ensure that the differential head between the landfill and the aquifer did not exceed the 1.27 m value used in this analysis). It is likely that an appropriately engineered and maintained underdrain system could function adequately for 20 years.

CONCLUSION

This paper has examined the effects of mass of contaminant, infiltration into a landfill, and contaminant transport pathway on the contaminating lifespan of a landfill. The effects of fracturing in the till beneath a landfill have been examined using an easy to use computer model to examine the implications of fracturing and to assess the effect of either a natural or man-made (e.g. compacted liner) intact layer in contact with the fractured till. The concept of developing "triggers" to initiate leachate control measures in the event of a failure of a leachate underdrain system has been discussed and the development of a trigger diagram has been demonstrated for a hypothetical case.

Based on the results of this study, it is concluded that even if the till beneath a proposed landfill site is fractured, with appropriate hydrogeologic and hydrogeotechnical investigation and design, it may often be possible to develop a landfill system which will not have an unacceptable impact on critical receptors such as an aquifer beneath the fractured till. However, consideration should be given to the potential for failure of the engineered system (especially the leachate underdrains) and the leachate levels and concentrations should be monitored for the entire contaminating lifespan of the landfill. If these levels exceed trigger values (which can be established as discussed in this paper) then appropriate leachate control measures should be initiated to control contaminant at the source (i.e. in the landfill) rather than waiting until a predictable, unacceptable impact occurs on groundwater resources.

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TABLE 1 SUMMARY OF PARAMETERS CONSIDERED

Quantity						
Length of Landfill L (m)	200					
Reference Height of Leachate H_r (m)	[a] 10	[b] 10	[c] 5	[d] 5		
Initial Concentration c_0 (mg/L)	[a] 1000	[b] 1000	[c] 1000	[d] 2000		
Downward Darcy Velocity v_a (m/a)	[0] 0	[1] 0.05	[2] 0.01	[3] 0.005	[4] 0.005	[5] 0.003
Thickness of Fractured Till H_T (m)	[0] 4	[1] 4	[2] 4	[3] 3	[4] 3	[5] 2
Porosity of Clay Matrix n_m (-)	0.4					
Hydraulic Conductivity of Fractured Till k_T (cm/s)	$1x10^{-7}$					
Thickness of Unfractured Till H_B (m)	[0] 0	[1] 0	[2] 0	[3] 0	[4] 1	[5] 1
Hydraulic Conductivity of Unfractured Till k_B (cm/s)	$2x10^{-8}$					
Thickness of Clay Liner H_L (m)	[0] 0	[1] 0	[2] 0	[3] 1	[4] 0	[5] 1
Hydraulic Conductivity of Liner k_L (cm/s)	$2x10^{-8}$					
Total Head Drop Between Base of Landfill and the Aquifer Δh (m)	[0] 0	[1] 6.35	[2] 1.27	[3] 1.27	[4] 1.27	[5] 1.27
Thickness of Underlying Aquifer h (m)	1					
Porosity of Aquifer n_b (-)	0.3					
Horizontal Darcy Velocity in Aquifer v_b (m/a)	$v_b = L.v_a$					
Diffusion Coefficient in Matrix D_m (m^2/a)	0.02					
Retardation Coefficient for Matrix R_m (-)	1					
Coefficient of Hydrodynamic Dispersion Along Fractures D (m^2/a)	0.06					
Fracture Spacing* $2H_1 = 2H_2$ (m)	1					
Fracture Opening Size* $2h_1 = 2h_2$ (μm)	10					

Note: Number in brackets [1] or [a] indicates the case; the number following the brackets gives the value of parameter used for that case. Where only one value is given, it is the value used for all cases.

* Assuming orthogonal fractures at equal spacings $2H_1$ and $2H_2$ and with equal fracture opening sizes $2h_1$ and $2h_2$.