Contaminant impact assessment and the contaminating lifespan of landfills

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Some of the factors to be considered in performing impact assessments for proposed municipal and non-hazardous waste landfill sites are discussed. These factors include the effect of the mass of contaminant, infiltration, and attenuation in the hydrogeologic system on the contaminating lifespan of a landfill. The potential impact of fracturing of the soil separating the landfill from an underlying aquifer is examined. The influences of a compacted clay liner and (or) a natural, intact clayey layer below the fractured soil are 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 underdrain system in a landfill.

Key words: environmental impact, contaminant migration, landfill, design, fractures, liners, groundwater, leachate collection.

Certains des facteurs dont il faut prendre en considération dans l'évaluation environnementale des impacts de lieux d'enfouissement municipaux de déchets non toxiques sont discutés. Ces facteurs incluent l'effet de la masse de contaminants, de l'infiltration et de l'atténuation dans le système hydrogéologique sur la durée de contamination d'un lieu d'enfouissement. L'impact possible de la rupture du sol séparant le lieu d'enfouissement d'une formation aquifère sous-jacente est étudié. L'influence d'une crépine d'argile compactée ou d'une couche argileuse naturelle intacte sous le sol en rupture est examinée. Le concept d'élaboration de dispositifs de déclenchement de mesures de contrôle de l'écoulement souterrain et leur impact possible sur la nappe phréatique sont discutés en fonction de la durée de conception du système de drainage souterrain d'un lieu d'enfouissement.

Mots clés : impact environnemental, migration des contaminants, lieu d'enfouissement, conception, rupture, crépines, nappe phréatique, collecte des eaux d'écoulement.

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Introduction

Unweathered clayey soils have been considered to represent a relatively ideal environment for the location of waste disposal sites. It is well recognized that "weathered" soils are fractured. However, conventional hydrogeologic investigations of "unweathered" soils have typically implied that they are unfractured. Recent research and field investigations, which have included angled boreholes or deep test pits in clayey tills, suggest that conventional investigations may be misleading and that many of these unweathered soils 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, personal communication). Typically, the fracture frequency decreases with depth and, in most cases examined by the author, the fractures could be observed terminating at some depth within the test pit.

To maintain a balance between excavated soil and the soil used for berms and landfill cover, it is often desirable, from a civil engineering perspective, to locate the base of the landfill in the unweathered, but fractured, soil. These soils (e.g., tills) are frequently underlain by aquifers. This then raises the question as to what potential impact a waste disposal site, located in the fractured soil deposit, may have on an underlying groundwater resource.

The modelling of contaminant migration in fractured porous media has received considerable attention (e.g., see Neretnieks 1980; Grisak and Pickins 1980; Tang *et al.* 1981; Sudicky and Frind 1982; Baker 1982; Rowe 1988; Rowe and Booker 1989, 1990*b*) 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 contaminants from landfill sites through fractured tills and into underlying aquifers has not received much attention.

Recently, Rowe and Booker (1990*b*, *c*) 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 microcomputer, 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 the present paper is to discuss some of the factors to be considered in performing impact assessments for proposed landfill sites in fractured soil. Factors to be considered include the mass of contaminant and infiltration 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 influences of a compacted clay liner above and (or) a natural, intact clayey layer below the fractured soil will be illustrated with reference to a hypothetical landfill.

It is emphasized that this paper is concerned with the potential for migration of contaminants by advective–diffusive–dispersive transport. No consideration is given to movement of contaminant which is controlled by gravity (density); and, in particular, the migration of concentrated dense nonaqueous phase liquids is beyond the scope of this paper.

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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).

The contaminating lifespan of a landfill will depend on the mass of contaminant per unit area (i.e., the height of landfill), 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. For landfills with a leachate collection system which removes leachate for treatment, the greater the infiltration (and hence the volume of leachate generated), the shorter will be the contaminating lifespan, since there is greater opportunity for contaminant to be leached out and removed from the landfill. The greater the potential for attenuation along the escape pathway, the shorter the contaminating lifespan.

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 leachate collection and (or) contaminant migration will result in a decrease in leachate concentration with time.

Although the peak concentration, c_0, 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 (e.g., Cheremisinoff and Morresi 1976; Kirk and Law 1985; Hughes *et al.* 1971).

For the purposes of modelling the decrease in concentration in the leachate due to movement of contaminant into the collection system and (or) 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, which is given by

$$[1] \quad H_r = \frac{m_{TC}}{c_0 A_0}$$

where H_r = representative height of leachate, [L]; m_{TC} = total mass of a contaminant species of interest, [M]; c_0 = peak concentration of that species in the landfill, [ML^{-3}]; and A_0 = area through which contaminant can migrate into the underlying layer, [L^2]. It should be emphasized that H_r *is not* the actual height of leachate mounding. It is simply a convenient means of representing the mass of contaminant available for transport into the soil and (or) collection by the leachate collection system.

To illustrate the potential effect of finite mass and infiltration, consider the case of a conservative contaminant species, which is highly soluble and readily leached from the waste and does not decay due to biological activity in the landfill. If there is an infiltration, q_0, through the landfill cover and where all the leachate is collected (i.e., there is no migration into the underlying soils), it can be mathematically shown (see Appendix A) that the ongoing infiltration into the landfill gives rise to a decrease in the average leachate concentration with time (recognizing that leachate concentrations may vary seasonally) such that the concentration in the landfill at time t, $c_{0L}(t)$, is related to the peak concentration, c_0, by the relationship

$$[2a] \quad c_{0L}(t) = c_0 \exp\left\{\frac{-q_0 t}{H_r}\right\}$$

or, on rearranging terms, the time required for the leachate strength to reduce to some specified value, c_{0L}, is given by

$$[2b] \quad t = \frac{-H_r}{q_0} \ln\left\{\frac{c_{0L}}{c_0}\right\}$$

where H_r([L]) is the reference height of leachate and q_0 ([LT^{-1}]) the infiltration as previously defined.

As an example, consider a landfill where the average thickness of waste, H_w, is 10 m, the average dry density of that waste, ρ_{dw}, is 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_0, is given by $m_{TC}/A_0 = H_w \rho_{dw} p = 10 \times 500 \times 0.02 = 10$ 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 (m), is given by

$$H_r = \frac{m_{TC}}{A_0 c_0} = \frac{10}{1} = 10$$

Assuming an average infiltration through the landfill cover of 0.15 m/a, the decrease in chloride concentration with time, simply due to dilution in the landfill, can be calculated from [2*a*] as indicated for case *i* in Fig. 1. Thus, for this example, the chloride level would reduce to 250 mg/L after approximately 90 years.

If the infiltration was 0.3 m/a (all other factors being equal), the concentration would decay much faster (see case *ii*, Fig. 1) and would reduce from the peak value of 1000 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 $m_{TC}/A_0 = 0.001 \times 10 \times 500 = 5$ kg/m^2 and thus $H_r = 5$ m. For an infiltration of 0.15 m/a, this gives case *iii*, for which the results shown in Fig. 1 are precisely the same as those obtained for case *ii* (since the ratio H_r/q_0 is the same). Clearly, the same result would also be obtained for 5 m of waste if chloride represented 0.2% of the waste (since, again, $m_{TC}/A_0 = 5 \times 500 \times 0.002 = 5$ kg/m^2).

If one were to assume the same total mass of chloride, m_{TC}, as in case *i* above, but if the peak concentration, c_0, is 2000 mg/L (2 kg/m^3), then H_r (m) is given by

$$H_r = \frac{m_{TC}}{A_0 c_0} = \frac{10}{2} = 5$$

and the decrease in concentration with time given by [2*a*] for q_0 = 0.15 m/a is as shown by case *iv* in Fig. 1. In this case, the concentration decreases from the peak value of 2000 to 250 mg/L in about 70 years and to 125 mg/L in just over 90 years.

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

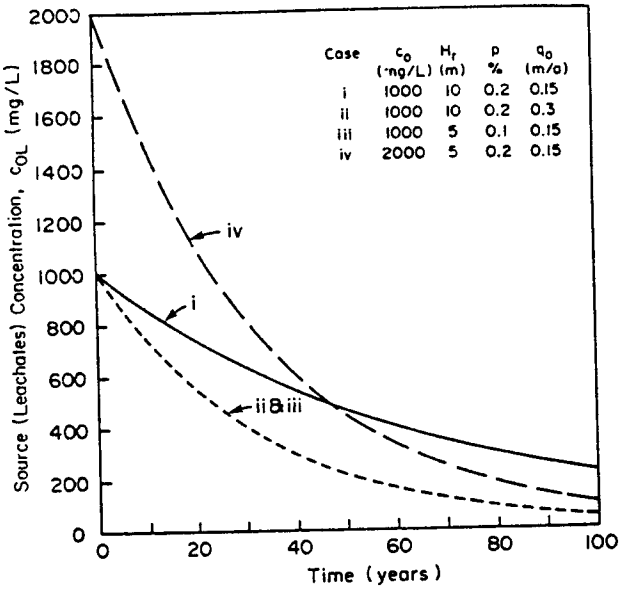

FIG. 1. Decrease in leachate (chloride) concentration with time.

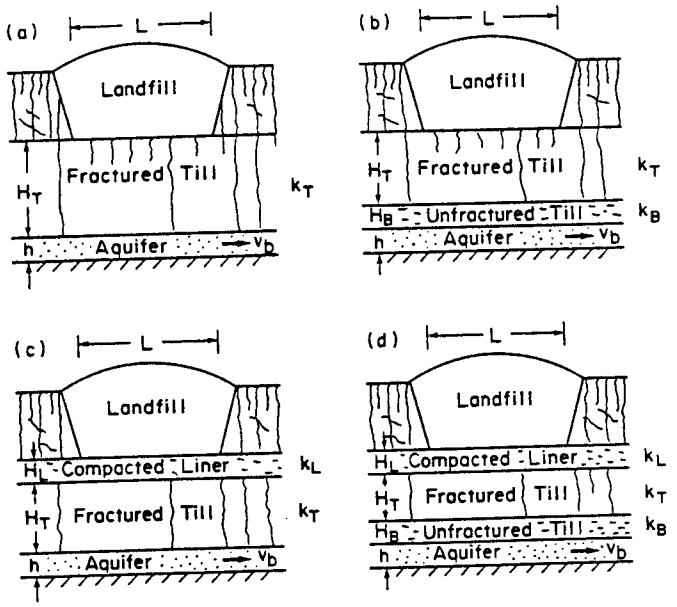

FIG. 2. Cases considered: (*a*) cases 0, 1, and 2; (*b*) case 4; (*c*) case 3; (*d*) case 5.

Environment has a "reasonable use" policy (MOE 1986). In the context of this policy, if a "reasonable use" for groundwater were decided to be as drinking water, an unacceptable impact could be interpreted as an increase in chloride at the site boundaries of 125 mg/L (or less if there are already background levels of chloride in the groundwater). In this case, for the examples considered in Fig. 1 it would be necessary for the leachate collection system to operate for between a maximum of about 140 years for case *i* and a minimum of about 70 years for cases *ii* and *iii* before dilution of the leachate would reduce chloride to levels that are sufficiently low that they would not have an unacceptable impact if they were discharged to the environment after failure of the collection system. Equation [2*a*] only considers decreases in concentration due to ongoing infiltration into the landfill. Clearly, if the contaminant experiences other decay mechanisms (e.g., biological decay), then the rate of decrease with time may be faster than that implied by [2*a*]. Furthermore, in reality, contaminants will generally have to pass through some form of hydrogeologic "barrier" 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 soil, and the properties of the underlying aquifer. Of these, the most important are the properties of the soil (assumed here to be till) and the head difference between the landfill and the aquifer. The implications of fracturing of the soil 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 Fig. 2. Typically, two hydrogeologic environments may be encountered. Figure 2*a* shows a landfill separated from an aquifer by a thickness, H_T, of fractured till, which has a bulk hydraulic conductivity k_T. 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_T, 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_B and hydraulic conductivity k_B, 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 Figs. 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 Fig. 2 have the properties defined in Table 1. For the purpose of calculating the Darcy velocity, the harmonic mean, $\bar{k}$, of hydraulic conductivities of the various units can be calculated from [3],

$$[3] \qquad \frac{H_L + H_T + H_B}{\bar{k}} = \frac{H_L}{k_L} + \frac{H_T}{k_T} + \frac{H_B}{k_B}$$

and the hydraulic gradient, i, is taken as the total head drop, Δh, between the base of the landfill and the aquifer, divided by the thickness, $H_L + H_T + H_B$,

$$[4] \qquad i = \frac{\Delta h}{H_L + H_T + H_B}$$

and hence the Darcy velocity (flux), v_a, is given by

$$[5a] \qquad v_a = \bar{k}i$$

That is,

$$[5b] \qquad v_a = \Delta h \Big/ \left(\frac{H_L}{k_L} + \frac{H_T}{k_T} + \frac{H_B}{k_B}\right)$$

TABLE 1. Summary of parameters considered

Parameter	Quantity
Length of landfill, L (m)	200
Reference height of leachate, H_r (m)	[i] 10; [ii] 10; [iii] 5; [iv] 5
Initial concentration, c_0 (mg/L)	[i] 1000; [ii] 1000; [iii] 1000; [iv] 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)	1×10^{-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)	2×10^{-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)	2×10^{-8}
Total head drop between bases 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)	$v_b = Lv_a$
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 (unless otherwise noted)
Fracture opening size* $2h_1 = 2h_2$ (μm)	10 (unless otherwise noted)

NOTE: Number in brackets [1] or [i] 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 $2h_2$.

In the natural setting (i.e., before construction of a landfill), there could be either upward or downward hydraulic gradients from the aquifer to the groundwater table. However, 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 groundwater 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 diffusion alone. 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 *i* (i.e., H_r = 10 m; c_0 = 1000 mg/L; q_0 = 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 Fig. 2*a*; see Table 1, cases 0 and *i* 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 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 and Booker 1983). From the results for case 0*i*, shown in Fig. 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 about 10% (i.e., 100 mg/L) is reached after about 275 years. Assuming that the initial background concentration of chloride in the aquifer is negligible, 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_{0L}, is less than the maximum increase in chloride permitted in the aquifer; this time can be calculated using [2*b*],

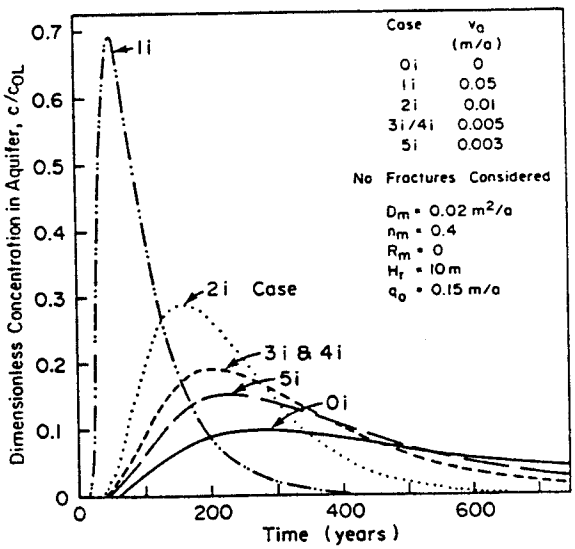

FIG. 3. Variation in chloride concentration in the aquifer, assuming migration through the matrix of the till (neglecting fractures).

viz., if $c_{0L}/c_0 = 0.125$, $H_r = 10$ m, and $q_0 = 0.15$ m/a, then $t = -H_r/q_0 \ln(c/c_0) = 139$ years), which implies a contaminating lifespan of about 140 years.

The question which then arises is 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 underlying till might have.

Lechate monitoring and "trigger" levels

Contingency measures for landfills are typically designed to remove contaminants if they "unpredictably" make their way through to an underlying aquifer (e.g., by installing purge wells). 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 the 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 the types of systems currently being used. Under these circumstances, it would be appropriate to anticipate that a failure of the underdrain system could 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 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 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.). Calculations can be performed 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

Suppose that a landfill is designed to operate with a hydraulic trap and that under these circumstances the impact is acceptable. Suppose that the hydraulic trap operates for a period of time, t_0, 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 can be modelled using the program POLLUTE v5). However, for simplicity, the change in velocity is considered here to occur quickly at time t_0, which approximately corresponds to the mean time between the time when the failure began to occur and the time when the full downward gradient is developed.

Once a downward hydraulic gradient develops, the hyraulic conductivity and fracturing of the underlying till become critical. Suppose, for the sake of discussion, that the bulk hydraulic conductivity of the 4 m ($H_T = 4$ m) of fractured till, k_T, 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. For determining the hydraulic 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). Thus the gradient, $i = \Delta h/H_T$, and the Darcy velocity (flux) can be determined ([5*a*]) to be $v_a = 0.05$ m/a.

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 the program POLLUTE v5 (op. cit.). 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 (1990*b*) and the program POLLUTE v5. Many situations will lie between these bounding cases, but by modelling these cases, it is possible to obtain a reasonable 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 (1990*b*). For the failure cases, the time shown in the figures (e.g., Figs. 3 and 4) represents the time after failure (i.e., after time t_0 as defined earlier).

Case 1*i* corresponds to a major failure of the leachate collection system for a landfill underlain by 4 m of fractured material (see Fig. 2*a* and Table 1). Assuming migration through the matrix of the till, Fig. 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), Fig. 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 1*i* in Figs. 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). As will be demonstrated later, the spacing of frac-

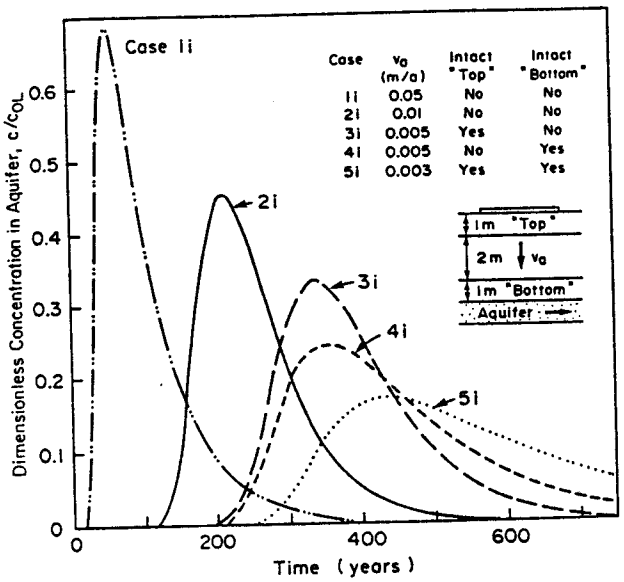

FIG. 4. Variation in chloride concentration in the aquifer, assuming migration along fractures.

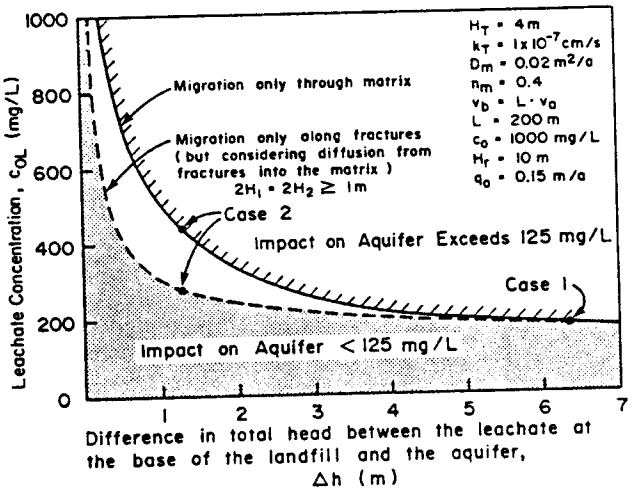

FIG. 5. "Trigger" concentration levels for leachate pumping at various levels of mounding for the hypothetical case being considered.

tures 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 1*i* can be used in conjunction with [2*b*] (or Fig. 1) to estimate how long the leachate collection system would have to work before a failure, and mounding to the level implied by case 1*i* could be allowed to occur. For example, taking the allowable increase in chloride concentration to be 125 mg/L and an attenuation factor of 0.69 (based on Figs. 3 and 4), the allowable value in the leachate at the time of failure would be $c_{0L} = 125/0.69 = 181$ 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 measures would be required. Thus case 1*i* corresponds to a differential head, Δh, of 6.35 m and a maximum allowable concentration in the leachate of 181 mg/L. This is plotted in Fig. 5 (which will be discussed subsequently). Assuming landfill condition *i* ($c_0 = 1000$ mg/L, $H_r = 10$ m, $q_0 = 0.15$ m/a), the time (in years) at which this failure could occur can be estimated from [2*b*], viz.,

$$t = \frac{-H_r}{q_0} \ln\left(\frac{c_{0L}(t)}{c_0}\right) = \frac{-10}{0.15} \ln\left(\frac{181}{1000}\right) = 114$$

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. Therefore, 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 Fig. 2*a*, it is assumed in case 2*i* 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 1*i*); 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 1*i*. Referring to Fig. 3, it is seen that for migration purely through the matrix, the contaminant reaches the aquifer at the 1% level within 45 years and the concentration increases to a peak value of about 0.29 c_{0L} of the leachate concentration at failure about 160 years after failure of the leachate collection system.

From Fig. 4, it is seen that migration through the fractures (but considering attenuation due to matrix diffusion) gives a much later first arrival in the aquifer at the 1% level (after about 125 years), but in this case the peak impact is 0.45 c_{0L} 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 bounds on the peak impact for the combination of parameters associated with case 2*i* are 0.29 (Fig. 3) and 0.45 (Fig. 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 Fig. 4, or less than 430 mg/L based on Fig. 3. These are plotted for $\Delta h = 1.27$ m in Fig. 5. Using [2*b*] or Fig. 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 solution would be required. By repeating the calculations discussed above for different assumed levels of leachate mounding, an envelope of trigger conditions can be constructed as shown in Fig. 5. By monitoring the leachate levels and concentrations and comparing them with the results shown in Fig. 5, it would be possible to determine whether supplementary leachate control (e.g., leachate wells) would be required for the case being considered. If the combination of leachate mounding and concentration plots below the dotted line, then the impact on the aquifer is expected

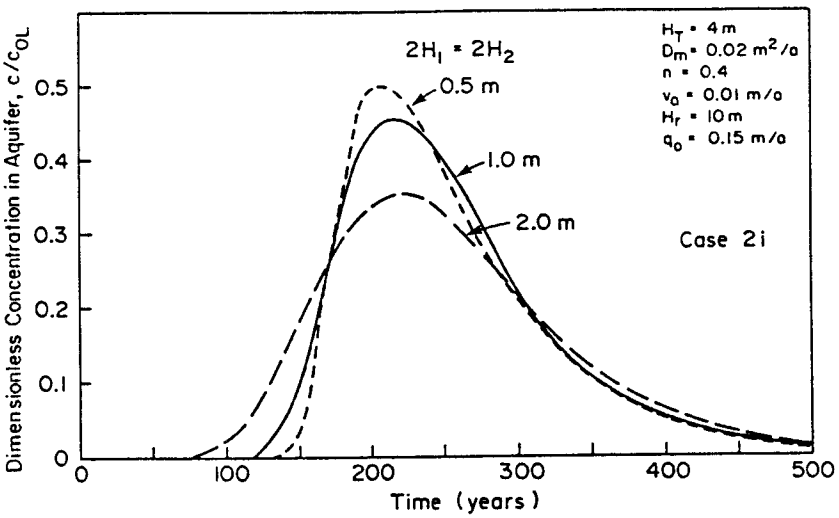

FIG. 6. Effect of fracture spacing on calculated impact in the aquifer. Case 2 — migration along fractures with attenuation due to matrix diffusion.

to be less than 125 mg/L and to be acceptable. 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 the 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 fracture spacing

The results presented in Figs. 4 and 5 assumed a fracture spacing of 1 m. However, fracture spacing will influence the potential impact as indicated in Fig. 6, which shows the variation in aquifer concentration with time for case 2*i* for three different fracture spacings. Decreasing the fracture spacing increases the time for first impact, but also increases the magnitude of the peak impact. This phenomenon is related to the amount of diffusion into the matrix, which can occur for different fracture spacings, and is discussed in more detail by Rowe and Booker (1989, 1990*a*). The important point to note here is that varying the fracture spacing can affect the calculated impact and the "trigger conditions" for leachate control. Since there will always be some uncertainty concerning the actual fracture spacing, it is appropriate to perform sensitivity studies over a reasonable range of uncertainty and to use the (conservative) envelope of results for impact assessment and for establishing trigger criteria.

Effect of a compacted clay liner over 3 m of fractured till

The scenario studied in the previous section involved 4 m of fractured till with a bulk hydraulic conductivity of 1×10^{-7} cm/s. The attenuation characteristics of this deposit could be improved by removing the top 1 m of the fractured till and replacing it with a compacted clay liner, assumed here to have a hydraulic conductivity of 2×10^{-8} cm/s. This corresponds to the situation shown in Figs. 2*c* and 2*d*. Assuming the same level of leachate mounding as in case 2*i*, the results for case 3*i* shown in Figs. 3 and 4 indicate that the clayey liner increases the time to first arrival in the aquifer and the time to peak impact, but reduces the magnitude of the peak impact. The peak impact evident from Figs. 3 and 4 was $0.19c_{0L}$ and $0.33c_{0L}$ respectively.

The results presented in Fig. 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 Fig. 3 than those in Fig. 4 for this case.

To keep the impact on the underlying aquifer to less than 125 mg/L, the results presented in Figs. 3 and 4 for case 3*i* combined with [2*b*] (or Fig. 1) suggest that it would be necessary for the leachate underdrain system to function between 30 and 65 years. Under these circumstances, the impact would then be acceptable, 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

In many situations the unweathered fractured till may be underlain by an unfractured till or clay layer as indicated in Fig. 2*b*. To illustrate this, case 4*i* assumes that this lower layer is 1 m thick with a hydraulic conductivity of 2×10^{-8} cm/s. It happens that the Darcy flux through this system is identical to that for case 3 with a 1 m thick compacted clay liner (with $k = 2 \times 10^{-8}$ cm/s). As a consequence, the results obtained, assuming migration through the matrix of the fractured till (as shown in Fig. 3), are identical to those obtained for case 3*i*. When migration through the fractures is considered, there is a difference between the results for cases 3*i* and 4*i*. It is evident from Fig. 4 for this specific case that it is more beneficial to have a 1 m thick natural intact layer at the bottom of the fractured till than to install a hydraulically similar compacted liner on top of the fractured till; however, this finding should not be extrapolated to other situations without direct verification. It is assumed in case 4*i* that when contaminant reaches

the bottom of the fractures, it will spread out and migrate evenly through the intact layer (as in case 3*i*, 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 Figs. 3 and 4, the peak impact for case 4*i* lies between $0.19c_{0L}$ and $0.24c_{0L}$ and, based on [4*b*] or Fig. 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 interest to assess the effect of removing and recompacting 1 m of fractured till beneath the base of the landfill, as shown in Fig. 2*d*. Analyses were performed assuming the parameters given for case 5*i* as given in Table 1. The results are shown in Figs. 3 and 4. Assuming that migration is only through the matrix (i.e., that the contaminant cannot 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 $0.15c_{0L}$ 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 Fig. 4), then the peak impact is slightly higher at $0.17c_{0L}$ about 430 years after failure of the underdrain. Although the magnitude of impact is very similar, the earlier impact time (Fig. 3) is likely to be more realistic for the situation shown in Fig. 2*d*, because the intact layer above and below the fractured zone is likely to force most of the contaminant migration to occur through the matrix of the fractured till, rather than through the fractures. 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.

At the outset of this paper, it was shown that the characteristics of the landfill, such as the height of waste, mass of a given contaminant, and the infiltration, would influence the decay of concentration in the landfill. As discussed, consideration of contaminating lifespan also involves consideration of the attenuation, which can occur between the landfill and any critical underlying receptor (e.g., an aquifer). The preceding discussion has focused on the assumed landfill conditions in case *i* as defined in Table 1. To illustrate the effect of other assumptions such as the mass of contaminant, the infiltration, and initial concentration, cases *ii*–*iv* were examined for the hydrogeological conditions of case 5 (considering migration through the fractures) and the results are shown in Fig. 7. Considering the peak impact for case *i* of about $0.17c_{0L}$, [2*b*] indicates that the leachate underdrain system would have to work for only about 20 years. Case *ii* involves the same mass of contaminant as case *i*, but assumes twice as much infiltration. Case *iii* involves only half the mass as in case *i* but assumes the same infiltration. Cases *ii* and *iii* give an identical response and a peak impact of less than $0.1c_{0L}$. Thus, either an increased infiltration or a decreased mass gives rise to a decreased impact (all other factors being equal). For these conditions, it would not be necessary to maintain the hydraulic trap, and even if the assumed "failure" mounding corresponding to a head drop, Δh, of 1.27 m developed immediately on completion of the landfill, the chloride impact on the aquifer would be less than 100 mg/L and, for these assumed conditions, would be acceptable.

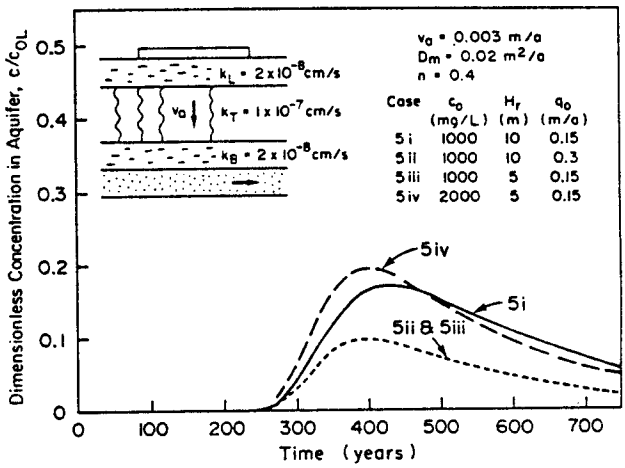

FIG. 7. Effect of landfill source characterization on calculated impact in the underlying aquifer. Case 5 — considering fractures.

Landfill condition *iv*, involving a peak concentration of 2000 mg/L (rather than 1000 mg/L assumed for the other cases) but the same total mass of chloride as for case *i*, gives a slightly larger peak impact of about $0.19c_{0L}$, and based on the parameters for this case, the leachate underdrain system would need to function for less than 40 years before mounding to give a Δh of 1.27 m could be permitted.

Conclusions

This paper has examined the effects of mass of contaminant, infiltration, 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 a 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 that 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 the established trigger values, appropriate leachate control measures should be initiated to control contaminant at the source (i.e., in the landfill), rather than waiting until a predictable, but unacceptable, impact occurs on groundwater resources.

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Appendix A

Assume that a landfill is at field capacity and that, after passing through the waste, the infiltration, q_0, is either collected by a leachate collection system or removed from the landfill by advective transport into the adjacent soil. As indicated by [1], the mass of contaminant initially in the landfill can be represented by the "representative height of leachate," H_r. Thus, neglecting diffusion into the underlying soil, a mass balance equation can be written for the landfill as follows:

[A1] mass of contaminant at time t

= initial mass of contaminant

− mass removed from landfill up to time t

which can be symbolically written as

$$[A2] \quad c_{0L}(t)H_rA_0 = c_0H_rA_0 - \int_0^t q_0A_0c_{0L}(\tau)\,d\tau$$

where $c_{0L}(t)$ is the concentration in the landfill at time t. Dividing throughout by H_rA_0 gives

$$[A3] \quad c_{0L}(t) = c_0 - \frac{q_0}{H_r}\int_0^t c_{0L}(\tau)\,d\tau$$

Using a Laplace transform, it is then a simple matter to obtain a solution to [A3], viz.,

$$[2a] \quad c_{0L}(t) = c_0 \exp\left\{\frac{-q_0t}{H_r}\right\}$$

Appendix B. Assumptions and features of program POLLUTE v5

The program POLLUTE v5 (Rowe and Booker 1983) which was used to obtain the results presented in this paper, is a semi-analytic finite layer program based on the theory proposed by Rowe and Booker (1985, 1987, 1990*b*, *c*). The basic features of the analysis are as follows:

- The soil deposit beneath the landfill is subdivided into layers. Each layer may have different properties (and may be fractured or unfractured), but the properties at any point in a given layer are the same.
- Migration through unfractured layers is governed by the one-dimensional advection–diffusion equation.
- Migration through fractured layers is governed by advective–dispersive transport along the fractures, coupled with matrix diffusion from the fracture into the porous matrix of the soil adjacent to the fractures. One-, two-, or three-dimensional orthogonal fracture networks can be considered; however, the fracture spacing in any one plane is uniform.
- The soil deposit may be underlain by either a relatively permeable or an impermeable base (e.g., a sand layer which

is permeable relative to the overlying clayey deposit or intact sound rock which is relatively impermeable relative to the overlying deposit); or the deposit may be of infinite extent (e.g., for modelling lateral migration).

- Provision is made for horizontal flow (and consequent dilution) within any permeable base stratum.
- The maximum concentration of contaminant in an underlying permeable stratum can be automatically determined.
- Provision is made for depletion of contaminant in the landfill with time as contaminant is removed by the leachate collection system and (or) by migration into the soil.
- Both linear and nonlinear (Langmuir or Freundlich) sorption can be considered.
- Radioactive (or other first-order) decay may be considered.
- Data preparation is minimal and, since it is a semi-analytic analysis, numerical error and computation time are very small in most cases.
- Both laboratory model tests and field situations can be modelled.