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ANALYSIS OF CONTAMINANT TRANSPORT THROUGH FRACTURED ROCK AT AN ONTARIO LANDFILL

R.K. ROWE AND J.R. BOOKER*
Geotechnical Research Centre
Faculty of Engineering Science
The University of Western Ontario
London, Canada N6A 5B9

*University of Sydney, Australia

ABSTRACT

The effects of fracture spacing, fracture opening size, Darcy velocity and dispersion upon the calculated contaminant plume in a fractured shale are examined. It is shown that the calculated contaminant plume, based on a reasonably hydrogeologic input, is consistent with the observed, very limited, extent of the contaminant plume at a 15 year old landfill in Burlington, Ontario. The results demonstrate that matrix diffusion can play a very significant role in the attenuation of contaminant migrating in fractured porous media.

INTRODUCTION

In the past, fractured rock has been considered, by some, to be a poor medium for the disposal of waste because of the potential for contaminant migration along fracture planes. This potential is well recognized. However, various investigators [1,7,10-12 and others] have also recognized the importance of matrix diffusion as an attenuation mechanism and have developed models which can consider its impact on contaminant migration in fractured porous media. The issue of contaminant migration in shale has assumed practical significance with respect to a recent proposal by the Region of Halton to site a municipal waste landfill on a fractured Queenston shale deposit in Burlington, Ontario. The proposed site is located directly between two existing landfills which have been constructed directly on the existing shale approximately 15 years ago ("Burlington" Landfill) and 30 years ago ("Bayview Park" Landfill). Neither landfill was originally constructed with a leachate collection system. A toe drain has been retrofitted to both landfills. The subsequent northern expansion of the Burlington landfill [4] is located directly on clayey till and has a leachate underdrain system. This expansion is not expected to have led to any significant leachate egress into the shale.

Rowe and Booker [11] recently published a technique for modelling contaminant migration in fractured media assuming parallel planar fractures and considering a finite source of contaminant. The objective of this paper is to describe the results obtained by applying this technique to predict the contaminant plumes that might be expected from the existing Burlington landfill under various assumed conditions, and to compare the calculated and observed migration downgradient of that landfill.

Basic Assumptions and Parameter Selection

The model proposed by Rowe and Booker [11] represents an extension of earlier formulations published by Barker [1] and by Sudicky and Frind [12]. It assumes

1. one dimensional, parallel periodic fractures of a spacing 2H;
2. the width (2b) of each fracture is much smaller than its length;
3. the permeability of the intervening porous matrix is low and that transport within the matrix is mainly by molecular diffusion;
4. transport along each fracture is much faster than transport within the matrix;
5. transverse diffusion and dispersion within each fracture assumes complete mixing across its width at all times; and
6. the source concentration varies as the mass of contaminant within the landfill decreases due to both the collection of leachate and the movement of contaminant into the rock.

Based on one interpretation of the available data, the lateral flow through the fractured rock would be expected to lie within the range 0.05-0.125 $m^3/m^2/a$ [6]. However, one investigator [2] has suggested that the flow could be substantially higher--perhaps up to 1.89 $m^3/m^2/a$. Thus, calculations were performed for a number of cases involving Darcy velocities of 0.05, 0.125, 0.7 and 1.89 m/a.

Although chloride is normally used as an indicator of the extent of contaminant migration from landfills, it does not provide a good leachate diagnostic in this case because the natural high salinity of the unpotable groundwater is such that the concentrations of chloride in the groundwater greatly exceed values in the leachate. Thus, due to matrix diffusion out of the fractures, the chloride concentration in exfiltrating groundwater (leachate) should actually increase with distance from the landfill rather than be attenuated.

The diffusion coefficient for chloride out of the shale was determined from a series of three tests to be 1.5×10^{-6} cm^2/s (0.0047 m^2/a). This was used as a reference value for estimating the tortuosity and hence the diffusion coefficients of other chemical species.

Based on the measured water content of the shale, the matrix porosity n_m of the shale was determined to be between 0.1 and 0.11.

For the purposes of calculating the expected extent of a contaminant plume due to the existing Burlington landfill, it was assumed that the leachate collection system at the north of the site collected all the contaminants from this portion of the site. Hence, the area of the landfill was taken to be the area of the portion which does not have a leachate underdrain system and this was approximated by a rectangular area of length L = 800 m and width W = 250 m.

Based on available data [8,13], the typical fracture spacing is between 5 cm and 30 cm, although in some locations spacings exceeding 1 m have been noted in the core logs. It may be expected that the open width of the significant fractures (i.e. the space available for uninhibited flow) will be more than 5 μm and typically less than 500 μm (0.5 mm). For a given fracture spacing and hydraulic conductivity, the fracture opening size can be estimated from Hoek and Bray [9]. As will be demonstrated, the fracture opening size is not a critical parameter and very similar results will be obtained for opening sizes varying by one or more orders of magnitude if the Darcy velocity remains constant.

There is some uncertainty as to the height of leachate which is applicable for the Burlington landfill because of uncertainty regarding (i) the initial mass of contaminants and (ii) the proportion of the infiltration which exfiltrates the landfill into the shale (since some may be collected by the toe drain). Based on the available data, the equivalent height of leachate H_{LF} [see 10, 11] was estimated to be $H_{LF} = 360/n_b$ (m). Since the analyses of the Burlington landfill are for relatively small times and since the major potential source of contaminant (i.e. the original portion of the landfill to the south of the site) has no leachate underdrain system beneath the waste, reasonable variations in the mass of contaminant used in the analysis (and hence the value of H_{LF}) does not affect the conclusions reached in the following section.

ANALYSIS OF MIGRATION FROM THE BURLINGTON LANDFILL

As previously noted, the normal inorganic species such as chloride cannot be used as a reliable indicator of the presence of contaminants at this site. As a consequence, recent investigations and monitoring have focused on organic contaminants [5].

The concentration of contaminant in a landfill will vary seasonally as well as over longer time spans. This natural variation is further complicated, when dealing with volatile organics, by potential volatilization of the organics after sampling and before testing. There is evidence, for example, that many of the leachate samples taken from the Burlington landfill were left too long before testing [3] and that as a consequence, the concentration levels monitored represent a minimum value; the actual concentration in the leachate may have been considerably higher. As a result, the selection of an appropriate source concentration is complicated. Assuming that the concentration of these organics varies in a log normal manner, there are three alternatives viz.

(i) the geometric mean of all the available data (1985-1987)
(ii) a representative "minimum" value based on the available data (1985-1987)
(iii) the geometric mean of good data (Oct.-Dec. 1987).

The geometric mean of all the data may be expected to underestimate the actual representative concentration since many of the data points may be biased by loss of contaminant prior to testing. The geometric mean of the "good" data may not be representative since it is based on only two data points. The representative minimum concentration can be regarded to be the very minimum concentration which could be attributed to the chemical species; the actual concentration would be higher than this value.

Since the leachate concentration data is for the 13-15 year period after construction of the landfill, the initial source concentration should be selected to give a corresponding source concentration after the elapsed period of time.

Monitoring downgradient of the Burlington landfill has involved the installation of two types of wells [5]. A number of the more recent wells (eg. GL 24, 26, 27) are constructed of stainless steel and were specifically intended for monitoring organics. Earlier wells (eg. GL 12, 16) are constructed of PVC and hence may, due to their construction, introduce some organic species to the water being sampled. As a consequence, data from these wells should be viewed with caution. All the following monitoring data is from the Gartner Lee 1987 monitoring report [5].

Four organic chemicals have been detected (at levels exceeding 2 µg/L) at the stainless steel well (GL 24) located approximately 25 m downgradient of the landfill [5]. These species and their appropriate concentrations are 1,1-dichloroethane (4 µg/L), vinyl chloride (12 µg/L), trichloroethylene (12µg/L)and cis-1,2-dichloroethylene (33 µg/L).

Two organic chemical species which have been detected (at above 2 µg/L) at GL16-IV (located about 95 m south of the landfill) are Trichloroethylene (5 µg/L) and cis-1,2-dichloroethylene (8 µg/L). While these concentrations could represent landfill derived contaminants, it is more likely that they arise from other sources. The reasons for this statement are twofold. Firstly,trichloroethyleneis present in the landfill at concentrations well below that of vinyl chloride and 1,1-dichloroethane and should have been far more retarded than either of these species. Thus one should ask why trichloroethylene has migrated 95 m to GL16-IV whereas vinyl chloride and 1,1-dichloroethane have not; this is not consistent with a plume having reached GL16. Secondly, the wells used at GL16 were constructed of PVC and may have been the source of this contaminant measured at this well.

Since space is limited, the following discussion will be focused on the migration of vinyl chloride and 1,1-dichloroethane since they are the most mobile of the four species and have very similar retardation coefficients.

Migration of Organics - 1,1-Dichloroethane

Figure 1 shows the calculated contaminant plume for 1,1-dichloroethane assuming a lateral flow (Darcy velocity) v_a of 1.89 m/a. Calculations performed with an initial source concentration of 217 µg/L give a source concentration (after 15 years) of 200 µg/L which corresponds to the geometric mean of the available leachate data [5]. If all concentrations (including the initial source) were doubled, this would give a source after 15 years equal to the geometric mean of the most reliable data (i.e. 400 µg/L). The initial source concentration of 54 µg/L gives the lowest expected concentration after 15 years of 48 µg/L. (The left and right axes of Fig. 1 correspond to the 217 µg/L and 54 µg/L source leachate concentrations respectively).

The location of four monitoring wells downgradient of the landfill is indicated on Fig. 1. As noted above, monitoring data has indicated that the concentration of 1,1-dichloroethane observed at GL24 was only 4 µg/L. None was detected at GL16-IV or GL27 (an apparently spurious result of 4 µg/L was reported at GL16-III but this is unlikely to be leachate derived).

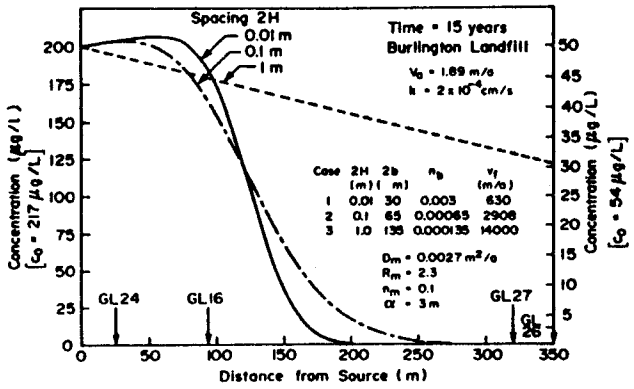

Figure 1. Calculated migration of 1,1-dichloroethane downgradient of the Burlington Landfill based on assumed k = $2x10^{-4}$ cm/s, i = 0.03, v_a = 1.89 m/a

Figure 1 gives theoretical solutions for three assumed fracture spacings (viz. 2H = 0.01 m, 0.1 m and 1 m). The fracture opening sizes were estimated from Hoek and Bray [9] based on the spacing between fractures an an assumed hydraulic conductivity of the rock of $2x10^{-4}$ cm/s (which corresponds to the Darcy velocity of 1.89 m/a and a gradient of 0.03). Examination of these three curves indicates that irrespective of the assumption regarding fracture spacing or the initial source concentration, the calculated curves based on a Darcy velocity of 1.89 m/a (k = $2x10^{-4}$ cm/s) are not consistent with field observations.

Figure 2 shows the calculated contaminant plumes assuming a Darcy velocity of 0.125 m/a (k~$1.3x10^{-5}$ cm/s) and three different fracture spacings. Examining these results in the context of the observed concentrations at GL24 and GL16, and allowing for the uncertainty as to the precise initial concentrations, it is evident that a Darcy velocity of 0.125 m/a would be consistent with the field observation for a fracture spacing of between 0.05 and 0.3 m. Since 70% of the rock core taken at this site had fractures in the range 0.05-0.3 m it is reasonable to consider that th Darcy velocity could be as high as 0.125 m. However, if the average fracture spacings between the landfill and GL24 were 0.3 m or greater then, th calculated impact is not consistent with field response and the actual Darcy velocity may be less than 0.125 m/a and could possibly be as low as 0.05 m/a.

Figure 3 shows the calculated increase in concentration with time at borehole GL24 (assuming a Darcy velocity of 0.125 m/a and three fracture spacings). It is evident that the time history of the concentration at this monitoring point would be of use in establishing the most appropriate parameters for describing migration in the fractured rock system. No 1,1-dichloroethane was detected at GL24 prior to November 1987 [7]. This woul tend to suggest that the concentration is only just starting to arrive at GL24 after about 15 years and would be most consistent with the results shown for case 4 (v_a = 0.125 m/a and 2H = 0.05 m). Additional monitoring will be useful in confirming the trends at GL24.

Based on the foregoing examination of 1,1-dichloroethane, one could tentatively conclude that the Darcy velocity downgradient of the Burlingto landfill is less than or equal to 0.125 m/a.

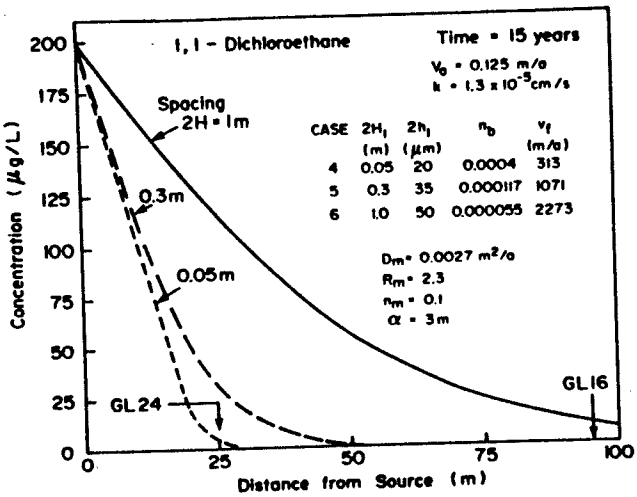

Figure 2. Calculated migration based on k = 1.3x10^{-5} cm/s, i = 0.03, v_a = 0.125 m/a

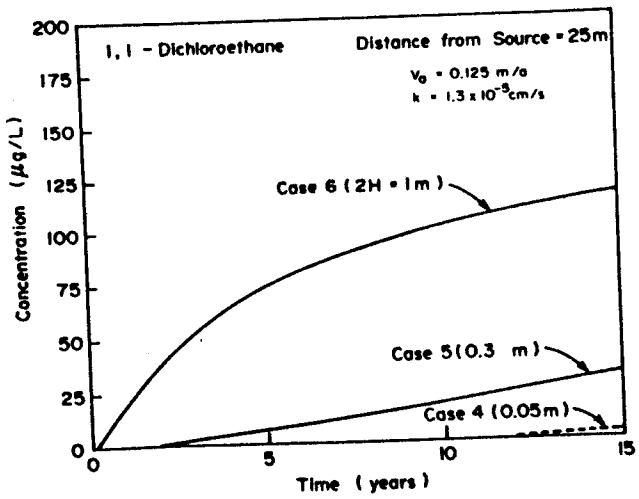

Figure 3. Calculated variation in concentration with time at monitoring point GL24, 25 m downgradient of the Burlington Landfill k = 1.3x10^{-5} cm/s; i = 0.03; v_a = 0.125 m/a

Vinylchloride

Vinylchloride has a retardation coefficient very similar to that of 1,1-Dichloroethane. Thus to sufficient accuracy, the results presented in Figures 2 and 3 may be regarded to be equally appropriate for vinylchloride provided that allowance is made for the difference in initial concentration. The current concentration of vinylchloride in the landfill has varied from 283 to 2010 µg/L with a geometric mean of 780 µg/L. The observable value of GL24 was 12 µg/L. This represents 1.5% of the current leachate concentration and is comparable with the level of 1,1-dichloroethane (1-2% of the leachate value) detected at GL24. No vinylchloride was detected prior to November 1987. Thus the migration of vinylchloride is entirely consistent with the migration of 1,1-dichloroethane and hence tends to confirm the results of 1,1-dichloroethane.

CONCLUSION

This paper has examined the effects of fracture spacing, Darcy velocity and dispersivity on the migration of contaminantsthrough fractured rock. The results of these analyses are compared with the observed data relating to the migration of contaminants from a municipal landfill located in Burlington, Ontario. It has been shown that the calculated contaminant plume based on a reasonable hydrogeologic evaluation of the site (which gives a Darcy velocity of between 0.05 and 0.125 m and average fracture spacing between 0.05 and 0.3 m) is consistent with the observed, very limited, extent of the contaminant plume. Based on these analyses, contaminant would only be expected to have migrated about 25 m in 15 years despite the fact that these conditions correspond to groundwater velocity ranging from 50 m/a to 1071 m/a. These results clearly demonstrate the significant role which matrix diffusion can have on the attenuation of contaminants migrating through fractured porous media.

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REFERENCES

1. Barker, J.A. Laplace transform solutions for solute transport in fissured aquifers. Advanced Water Research, 1982, 5(2), pp. 98-104.

2. Conestoga-Rovers & Associates. Evaluation of Halton Regional Landfill Technical Studies - Site F - Burlington: Hydrogeologic Report. Report 1354-10, Submitted to the City of Burlington, February 1988.

3. Feenstra, S. Personal Communication, 1988.

4. Gartner Lee and Associates. Hydrogeological Study: Burlington Landfill Site Continued Use for M.M. Dillon Ltd.), 1984, p. 0-31.

5. Gartner Lee Ltd. Halton Regional Landfill - Burlington 1987 Monitoring Report. Report GLL87-224, Submitted to the Region of Halton, March 1988.

6. Geocon. Review of Hydrogeological Report on Site F - Burlington, Proposed Landfill; Region of Halton. Report 43805, Submitted to David Estrin, Barristers and Solicitors for the Town of Milton, October 1987.

7. Grisak, G.E. and Pickens, J.F. Solute transport through fractured media, 1, The effect of matrix diffusion. Water Resources Research, 1980, 16(4), pp. 719-730.

8. Hewetson, J.P. An investigation of the groundwater zone in fractured shale at a landfill. M.E.Sc. Thesis, University of Waterloo, p. 27.

9. Hoek, E. and Bray, J.W. Rock Slope Engineering, 3rd Ed. The Institute of Mining and Metallurgy, London, p. 133.

10. Rowe, R.K. Contaminant migration through groundwater: The role of modelling in the design of barriers. Canadian Geotechnical Journal, 1988, Vol. 25, No. 4 (In Press).

11. Rowe, R.K. and Booker, J.R. Modelling of contaminant movement through fractured or jointed media with parallel fractures. Proceedings of 6th International Conference on Numerical Methods in Geomechanics, Innsbruck, April 1988, pp. 855-862.

12. Sudicky, E.A. and Friend, E.O. Contaminant transport in tractured porous media: Analytical solution for a system of parallel fractures. Water Resources Research, 1982, 18, pp. 1634-1642.

13. Trow Hydrology Consultants Ltd. Halton Regional Landfill Technical Report - Site F Burlington, Vols. 1, 2, 3. Submitted to the Region of Halton, Sept. 1986.