

AN EXAMINATION OF THE POTENTIAL IMPACT ON GROUNDWATER
QUALITY OF THE PROPOSED HALTON LANDFILL AT SITE F
IN THE CITY OF BURLINGTON, REGION OF HALTON

Prepared by: R. Kerry Rowe Ph.D., P.Eng.

For: David Estrin
Barristers & Solicitors
on behalf of the Town of Milton

October, 1987

TABLE OF CONTENTS

	Page
1.0 INTRODUCTION AND SUMMARY	1
1.1 Calculated Extent of Contaminant Migration From the Existing Burlington Landfill	5
1.2 Potential Impact of the Proposed Halton Landfill: Site F	7
1.3 Conclusion	9
2.0 TRANSPORT MECHANISMS AND ATTENUATION	10
3.0 MASS OF CONTAMINANT	19
4.0 PROBLEM IDEALIZATION	23
5.0 SELECTION OF PARAMETERS	26
5.1 Assumed Landfill Dimensions	26
5.2 Lateral Flow (Darcy Velocity) in the Deep Groundwater System	27
5.3 Fracture Spacing and Aperture Size	30
5.4 Diffusion Coefficients, Retardation Factors and Matrix Porosity	30
5.5 Dispersivity	32
5.6 Initial Concentration and Mass of Contaminant	32
6.0 ANALYSIS OF THE EXISTING BURLINGTON LANDFILL	33
6.1 Migration of Benzene and Similar Organics	33
6.2 Migration of Conservative Contaminants	39
6.3 Summary	43
7.0 ANALYSIS OF THE PROPOSED HALTON LANDFILL - SITE F	44
8.0 WORST CASE SCENARIO - SITE F	53
9.0 CONCLUSION	56
10.0 REFERENCES	61
APPENDIX A: Theory (Rowe & Booker, to be published)	63
APPENDIX B: Results of Additional Calculations Relating to the Burlington Landfill	68
APPENDIX C: Analysis of Leaching Profiles for Chloride at Site F	73

LIST OF FIGURES

Figure	Description	Page
1	Problem Idealization: Parallel Cracks	13
2	Illustration of the Effect of Matrix Diffusion	15
3	Projected Decline in Chloride Concentrations in Landfill Leachate	20
4	Calculated Migration of Benzene From the Burlington Landfill After 15 Years	36
5	Calculated Migration of "Chloride" From the Burlington Landfill After 15 Years	40
6	Calculated Migration of "Chloride" From the Burlington Landfill After 15 Years	41
7	Calculated Variation in "Chloride" Concentration With Time At a Point 300 m Downgradient of the Proposed Site F Halton Landfill	47
8	Calculated Variation in "Chloride" Concentration With Time At a Point 800 m Downgradient of the Proposed Halton Landfill (Site F)	48
9	Calculated Variation in "Benzene" Concentration With Time At a Point 300 m Downgradient of the Proposed Halton Landfill (Site F)	49
10	Calculated Variation in Organic Contaminant Concentration With Time At a Point 800 m Downgradient of the Proposed Halton Landfill (Site F)	50
11	Calculated Variation in "Chloride" Concentration With Time Downgradient of the Proposed Halton Landfill (Site F) - "Worst" Case Scenario	54
12	Calculated Variation in "Benzene" Concentration With Time Downgradient of the Proposed Halton Landfill (Site F) - "Worst" Case Scenario	55
13	Calculated Migration of "Benzene" From the Burlington Landfill After 10 Years	69
14	Calculated Migration of "Chloride" From the Burlington Landfill After 10 Years	70
15	Calculated Migration of "Benzene" From the Burlington Landfill After 25 Years	71

LIST OF FIGURES

Figure	Description	Page
16	Calculated Migration of "Chloride" From the Burlington Landfill After 25 Years	72
17	Calculated Variation in Chloride Concentration With Depth Based on Leaching of an Initially Uniform Profile by Downward Flow	76
18	Effect of Fracture Spacing and Porosity On Calculated Concentration Profile $v_a = 0.001$ m/a	77

LIST OF TABLES

Table	Description	Page
1	Cases Considered	34

computer
modelling
not done by
Trowat P.

- predictability
- complexity

1.0 INTRODUCTION AND SUMMARY

This report examines the potential for attenuation of contaminants entering the fractured rock beneath the proposed Region of Halton Landfill at Site F (see Drg. 5, Vol. 3, THCL report on Site F, Exhibit H42A) as a result of diffusion into the matrix of the shale adjacent to the fractures.

The report has been prepared using the results obtained from modelling of contaminant transport at Site F conducted by the writer, as part of a contract between the Town of Milton and The University of Western Ontario (the UWO study), and described in the following sections of this report. The cases examined in the UWO study were selected after consideration of the hydrogeological information and the engineering design documents contained in Trow Hydrology Consultants Ltd.'s (THCL) report Volumes 1, 2 and 3 (Exhibits H40, H42, H42A) relating to Site F. The hydrological information contained in THCL's report was supplemented by additional hydrogeological interpretation provided by Geocon.

The phenomenon of attenuation due to matrix diffusion is well recognized and was considered in the UWO study. Thus, this study involved modelling of contaminant migration along fractures (due to advective-diffusive-dispersive transport) together with diffusion of contaminant from the fractures and into the matrix of the rock. Possible sorption of contaminant was also considered where appropriate. The model satisfies conservation of mass requirements. It also automatically allows for variations in concentration in the source leachate as the mass of contaminant within the landfill decreases due to both the collection of leachate and the movement of contaminant down into the rock.

The model assumes parallel planar fractures. It is recognized that the real situation is more complicated and will involve dead end fractures,

constrictions (i.e. narrowing followed by widening of the fracture) and a tortuous flow path (eg. flow along a near horizontal fracture then up (or down) a vertical fracture and then along another near horizontal fracture). The net effect of this can be expected to be manifest as a dispersion or spreading of contaminants in excess of that calculated on the basis of advective-diffusive transport along the parallel fractures. There is no easy way to establish this dispersion coefficient without field data. Similarly, there is uncertainty concerning the value of the groundwater velocity and fracture geometry. Under these circumstances, it is not possible to make precise predictions of contaminant migration no matter how sophisticated the theoretical model used. [Fortunately, however, the Burlington and Bayview Park landfills provide an indication of the movement (or lack of movement) of contaminant in a hydrogeologic environment which is similar to that beneath the proposed Halton landfill at Site F.] In this situation, theoretical models can be particularly useful for examining the implications of different combinations of parameters and the choice of parameters can be tested by comparing the calculated and observed extent of migration from existing landfills. By using these parameters in an analysis of the proposed Halton landfill at Site F, it is then possible to form an engineering opinion as to the likely performance of the proposed landfill.

From the standpoint of modelling contaminant migration, the proposed Site D is at least as complex as Site F and there is uncertainty regarding the value of relevant parameters at both sites. In both cases, modelling using reasonable combinations of parameters will provide an indication of the potential contamination of the groundwater. At Site F there is the unique advantage of having the documented performance of the two existing landfills which can be used to calibrate the choice of parameters.

show where
p. tests
are
do know
p. p. p. p. p.
look
Don't
really
know

TRY
not do
modeling
at D

his view

Based on available data, lateral migration through the fractured rock was modelled using an average lateral flow per unit area (v_a) of 0.05 m/a. For a fracture porosity of 0.001 this corresponds to an average linearized groundwater velocity of 50 m/a and appears to be within the range of values reported by THCL (Vol. 1, p. 6). For the purposes of considering a "worst case" scenario in the event that the leachate collection tiles all failed it was very conservatively assumed that leachate mounding would develop to the top of the waste mound. This assumption is more severe and conservative than that made in the study of Site D even though, if anything, one would expect the height of mounding to be less at Site F than at Site D due to the sloping nature of the site. Based on this very conservative assumption, the lateral flow per unit area (Darcy velocity) in the deep rock system was estimated to increase from .05 m/a to 0.125 m/a. This increase in velocity corresponds to an increase in the lateral gradient by a factor of 2.5 due to the mounding and is based on the assumption that the head 300 m downgradient would not change due to mounding. If the head at this point did increase then the flow would be even smaller than indicated above. For a fracture porosity of 0.001, the flow of 0.125 m/a corresponds to an average linearized groundwater velocity (fracture velocity) of 125 m/a.

Calculations were also performed for a lateral flow per unit area (v_a) of 0.7 m/a. If one assumes that this flow passes through a thickness, T , of rock of 25 m then this flow corresponds to the infiltration of 14 mm which (based on their original computer model) THCL claim passes into the rock beneath Site F at present.

Consideration was given to the migration of both a conservative contaminant species (chloride) and a volatile priority pollutant (benzene).

The diffusion coefficient for chloride into the shale at Site F was

Cross
re weakly
fract. controlling
layer

not have
much
impact on
out flow

determined from a series of three tests to be $1.5 \times 10^{-6} \text{ cm}^2/\text{s}$ ($0.0047 \text{ m}^2/\text{a}$). Attenuation due to matrix diffusion will be greatest when the matrix diffusion coefficient is high and so for this type of modelling it is conservative to adopt a low value of diffusion coefficient (the opposite is true when one deals with diffusion through an unfractured medium). To ensure that the calculations were conservative, the matrix diffusion coefficient for chloride was reduced by a factor of 2.5 and the value used in the study was $0.0019 \text{ m}^2/\text{a}$ (i.e. $6 \times 10^{-7} \text{ cm}^2/\text{s}$). This is very close to the value of $5 \times 10^{-7} \text{ cm}^2/\text{s}$ used by THCL in their modeling of diffusion through unfractured rock.

The matrix diffusion coefficient for benzene was determined by multiplying the diffusion coefficient in free solution by the tortuosity factor of 0.047 giving a value of $0.0011 \text{ m}^2/\text{a}$ (i.e. $3.6 \times 10^{-7} \text{ cm}^2/\text{s}$). This is also considered to be a conservative value. The retardation factor for benzene was estimated to be $R_m = 5.5$.

Analyses performed for a very wide range of fracture aperture widths (from $5 \text{ }\mu\text{m}$ to $500 \text{ }\mu\text{m}$) and porosities ranging from 0.0001 to 0.01 indicated that to all practical purposes the fracture opening size and fracture porosity has no effect on the contaminant plume for a given flow per unit area through the fracture system. With regard to fracture spacing, for small times (i.e. times of the order of tens of years), the fracture spacing does have an effect on the extent of the contaminant plume. Fracture spacing (within the range typical in the environs of Site F) ceases to be of significance for times of the order of hundreds to thousands of years.

The most important parameter controlling the migration of contaminants in a fractured rock system is the average flow (per unit area) which can be expressed

in terms of the Darcy velocity, v_a . The smaller the flow (Darcy velocity), the greater the level of attenuation (all other factors being equal) because attenuation due to matrix diffusion is dependent on the total mass of contaminant being transported along the fractures rather than the average linearized groundwater velocity.

1.1 Calculated Extent of Contaminant Migration From the Existing Burlington Landfill

The extent of the contaminant plume from the existing Burlington landfill was calculated for a range of assumed lateral flows (Darcy velocities) within the rock:

(i) Calculations were performed for the expected conditions in the environs of Site F, assuming that leachate mounding in the portion of the Burlington landfill which does not have a leachate underdrain system does not significantly increase the Darcy velocity (which was taken to be 0.05 m/a). Under these conditions, it is considered unlikely that benzene would be detected in any significant quantity at any of the Gartner Lee and Associates Limited (GLAL) boreholes southeast of the landfill after only 15 years migration through the rock.

(ii) Analyses were performed assuming that the "worst case" scenario proposed for Site F* is applicable to the portion of the existing Burlington landfill which does not have leachate underdrain systems (i.e. taking the Darcy velocity to be 0.125 m/a). For this case, calculations of the expected concentrations of benzene at points 25 m downgradient from the landfill (e.g. at boreholes GL 12 and 24) indicate that after 15 years migration:

- benzene may be detected at levels of up to 15% of the source concentration if the average fracture spacing was about 30 cm; however,
- benzene would not be detected at levels exceeding $1 \mu\text{g/L}$ if the average fracture spacing was 5 cm.

* For the event where there is failure of the leachate collection system and leachate mounding to the top of the waste.

see
ws re
K + gradient

x2 1/2

≡

20 ug/L

Does this
have a
large
impact

(iii) Analyses performed assuming a Darcy velocity of 0.7 m/a (which is related to THCL's calculated infiltration to the deep groundwater system in their original computer flow model), indicate that substantial quantities of benzene and other less reactive contaminants should be detected at 25 m from the landfill (e.g. at borehole GL 24). There should be no problem in identifying the presence of a contaminant plume if the Darcy velocity was of this magnitude.

To be consistent with THCL's attempt to provide an indication of the potential migration of conservative species of contaminant, analyses were also performed for chloride. Chloride was used as representative of highly mobile non-reactive contaminants; however, the following comments should not be regarded as being directly applicable to chloride itself because the very poor quality of groundwater in this area involves concentrations of chloride in the rock which greatly exceed values expected in the leachate. Thus, due to matrix diffusion out of the fractures, the chloride concentration in infiltrating groundwater or leachate should actually increase with distance from the landfill rather than be attenuated.

Analyses of the migration of conservative contaminants from the existing Burlington landfill at the expected average lateral flow (per unit area) of 0.05 m/a indicate that the conservative ^{chlor}contaminant might be detectable at low concentrations 25 m from the landfill and not detectable at boreholes which are located further from the landfill.

Applying the "worst case" scenario for Site F to the existing Burlington landfill (i.e. adopting an average lateral flow of 0.125 m/a), calculations indicate that a conservative species such as chloride would be readily detectable 25 m from the landfill (i.e. at GL 12 and 24) and might be detected at low

concentrations at borehole GL 16 if the average fracture spacing is approximately 30 cm. For fracture spacing substantially less than 30 cm (e.g. 5 cm), conservative contaminants would probably not be detectable at GL 15 or 16 after 15 years migration.

Calculations for the migration of conservative species for an average lateral flow of 0.7 m/a indicate that high concentrations of contaminant (exceeding 50% of the source concentration) would be expected at distances of between 75 m and 180 m from the landfill. Any reasonable combination of fracture spacing, fracture aperture size, fracture porosity, or dispersivity would be expected to show a similar trend.

The fact that these high levels of conservative contaminants are not observed in boreholes GL 12, 24 and 15 is considered to be good evidence that the Darcy velocity of contaminant transport is much smaller than the 0.7 m/a adopted in these calculations and hence the average recharge of the deep groundwater system is expected to be considerably less than 14 mm/a that THCL calculated assuming the existence of a distinct highly fractured unit.

The results of the Burlington Landfill Plume Delineation Study (GLAL, 1986) are most consistent with calculations performed using an average lateral flow per unit area of 0.05 m/a.

*Did this do
chlorides*

1.2 Potential Impact of the Proposed Halton Landfill: Site F

Analyses performed to assess the potential impact of the proposed Halton Landfill at Site F indicate that for an average lateral flow of about 0.05 m/a it

would take conservative contaminants about 200 years to reach a concentration of $0.01 c_0$ (i.e. 1% of the initial concentration) at a point 300 m downgradient of the landfill. The peak concentration calculated for this point was less than 7% of the leachate concentration (i.e. about 100 mg/L assuming an initial concentration $c_0 = 1500$ mg/L) and did not occur until 450 years after migration began. At a point 800 m from the landfill the peak concentration was only 4% of the leachate concentration and was not reached until about 1250 years after migration commenced.

Calculations performed for organic species such as benzene indicate that retardation due to sorption reduces the magnitude of the peak concentration and substantially increases the time to reach this peak. For example, 300 m from the proposed landfill the peak concentration of benzene was calculated to be about $0.015 c_0$ (i.e. 1.5% of the initial concentration) and this would not be reached until 2200 years after migration begins. At a point 800 m from the landfill the peak is even smaller and is not reached until 6000 years after landfilling (assuming conditions remain the same for this period of time).

Considering the worst case scenario of mounding of leachate to the top of the waste, calculations indicate that it would take a conservative species, such as chloride, about 200 years to reach a peak concentration of $0.15 c_0$ (15% of initial concentration) at a point 300 m downgradient. At a point 800 m away (near the CNR track) the peak concentration of $0.1 c_0$ (10% of c_0) would not be reached until about 525 years after failure of the collection system. The migration of retarded species, such as benzene, is even slower and at 300 m downgradient, the peak concentration of about $0.03 c_0$ (3% of c_0) would not be reached until about 900 years after failure of the collection system.

1.3 Conclusion

The contaminant studies reported herein have not taken account of the beneficial effects of dilution of contaminant due to recharge of the rock from outside of the proposed Site F. Nor have the potentially beneficial effects of attenuation mechanisms such as biodegradation and microbial transformation of organic contaminants discussed by THCL (Vol. 1, pp. 85-86) been considered. Thus the concentrations of contaminant calculated in this study are considered to be conservative.

The findings of the Burlington Landfill Plume Delineation Study (GLAL, 1986) are consistent with the results of analyses performed in the UWO study for an anticipated average lateral flow per unit area of 0.05 m/a (which corresponds to a fracture velocity of 50 m/a for a fracture porosity of 0.001). Based on the results of similar analyses performed for the proposed Halton landfill at Site F using the same average lateral flow per unit area of 0.05 m/a, it is concluded that there would be minimal effect on the groundwater quality southeast of Highway 403 due to contaminant migration in the deep groundwater system from Site F.

Even if one adopts the worst case scenario for the extreme situation of leachate mounding to the top of the waste, the potential impact on groundwater quality southeast of Highway 403 is relatively small.

As noted above, the proposed landfill will have minimal impact on deep groundwater quality. Thus, in the event that there was a discharge of this groundwater into the shallow groundwater system southeast of Highway 403, the impact of the landfill on this shallow groundwater would also be minimal. Furthermore, upward movement of deep groundwater through the till would result in additional attenuation and improvement in the deep groundwater quality. Indeed, in the event that the deep groundwater did discharge in this manner, one would

expect the impact of the naturally existing poor quality of this groundwater to be far more significant than any impact due to contaminant in this groundwater which originated from the proposed Halton landfill.

Comparing the results of analyses for both Sites D and F, it is concluded that the potential for significant groundwater contamination southeast of Highway 403 due to contaminant migration from the proposed Halton Landfill at Site F is minimal compared to the likelihood of contamination of aquifers and residential water supplies adjacent to the proposed Site D.

The analyses performed in this study and the conclusions reached are based on the assumption that the Proposed Halton Landfill at either Site F or D is for the disposal of municipal waste. No consideration has been given to the implications of disposing of quantities of hazardous/toxic waste over and above the levels normally encountered in municipal leachate.

2.0 TRANSPORT MECHANISMS AND ATTENUATION

As noted by THCL (Vol. 1, p. 76):

The assessment of the environmental impact of leachate from a municipal waste landfill on groundwater requires an adequate understanding of the migration of potential contaminants in the groundwater. Mechanisms including advection (flow of groundwater), diffusion and hydrodynamic dispersion will act to transport contaminants in groundwater. At the same time, mechanisms such as adsorption, precipitation, volatilization, biodegradation and matrix diffusion will act to attenuate or retard the transport of contaminants.

In a fractured rock flow system, the primary transport mechanism along the fractures will be advection (i.e. the movement of contaminants due to the flow of groundwater). In their report, THCL (Vol. 1, p. 76) comment that:

For contaminants which do not adsorb significantly on the subsurface soils or rock (non-reactive contaminants) or are not significantly removed from the groundwater by precipitation, volatilization or biodegradation reactions (conservative contaminants), migration occurs at the rate of the average linear groundwater velocity.

In fractured rock (or soil), the average linearized groundwater velocity v_f is related to the Darcy velocity v_a (i.e. flow per unit area) by the relationship

$$v_f = v_a / n_b$$

1 SA m of soil

k - 5
i 1/2

DV 2 1/2 m/sec

- 2 1/2 cu m of water.
not how long just av
need porosity
(1)

where n_b is the effective porosity through which flow is occurring. Thus, the fracture velocity v_f may be quite high because the effective porosity of the rock mass n_b may be quite small (THCL, Vol. 1, p. 78 quote a range between 0.1 to 0.001% based on hydraulic conductivity-fracture opening size-fracture frequency relationships). If conservative contaminants were to migrate along these fractures at the average linear groundwater velocity v_f , then the rate of contaminant migration through the rock from the existing Bayview Park and Burlington landfills would be very fast and there should be little difficulty in detecting significant quantities of contaminant in the groundwater at points well away from the landfill. For example, if the average linear groundwater velocity in the fractures downgradient of the existing Burlington landfill were 50 m/a, then 15 years after leachate enters the rock, contaminants should be readily detected at distances of up to 750 m. However, the available field data (eg. the Burlington Landfill Plume Delineation Study, GLAL 1986) suggests that this is not the case.

While it may be true that migration of conservative contaminants could occur at the rate of the average linear groundwater velocity in porous media where the matrix of the media (eg. the sand grains in a sandy aquifer) has a negligible effective porosity, it is not true that conservative contaminants (eg. chloride)

will migrate at the rate of the average linear groundwater velocity in fractured rock (or soil) systems where the matrix of the rock (or soil) between fractures has a significant effective porosity (e.g. see Grisak and Pickens, 1980). This is the case at Site F where the Queenston shale has a matrix porosity of approximately 10%. In cases such as this, diffusion of contaminants from fractures or fissures into the adjacent matrix can control the migration of contaminants and represents an important attenuation mechanism.

(The phenomenon of attenuation due to matrix diffusion is well recognized (eg. Barker and Foster, 1981; Foster, 1975; Freeze and Cherry, 1979; Grisak and Pickens, 1980; Grisak et al., 1980; Sudicky and Frind, 1982; Tang et al., 1981). THCL also recognize the importance of matrix diffusion and discuss this mechanism (pages 86-88, Vol. 1) using a diagram from Freeze and Cherry (1979) to illustrate the concept (reproduced here as Fig. 2). However, in their evaluation of contaminant migration (pp. 100-122), THCL adopt a simple theoretical model (the Ogata-Banks equation) which can not take account of attenuation due to matrix diffusion. Since a primary objective of the UWO study was to examine the effect of matrix diffusion on potential contaminant migration at Site F, this mechanism will be discussed in a little more detail in the following paragraphs.]

Figure 1 shows an idealization of a fractured system in which a series of parallel fractures with an aperture width of $2b$ are at an average spacing $2H$. If the matrix material on either side of the fracture was non-porous then contaminants would move along the fracture at the average groundwater velocity v_f and after some time t_1 the contaminant plume would extend a distance $x_1 = t_1 \cdot v_f$. For a constant source concentration c_0 , the concentration of contaminant in the fracture would be equal to c_0 up to the distance x_1 and would be at background concentration beyond this point. If one also considers diffusion within the

Leachate
Source :

Initial
Concentration
 $c_{LF} = c_0$

Concentration, c_{LF} ,
may vary as
contaminant is
transported into
the rock.

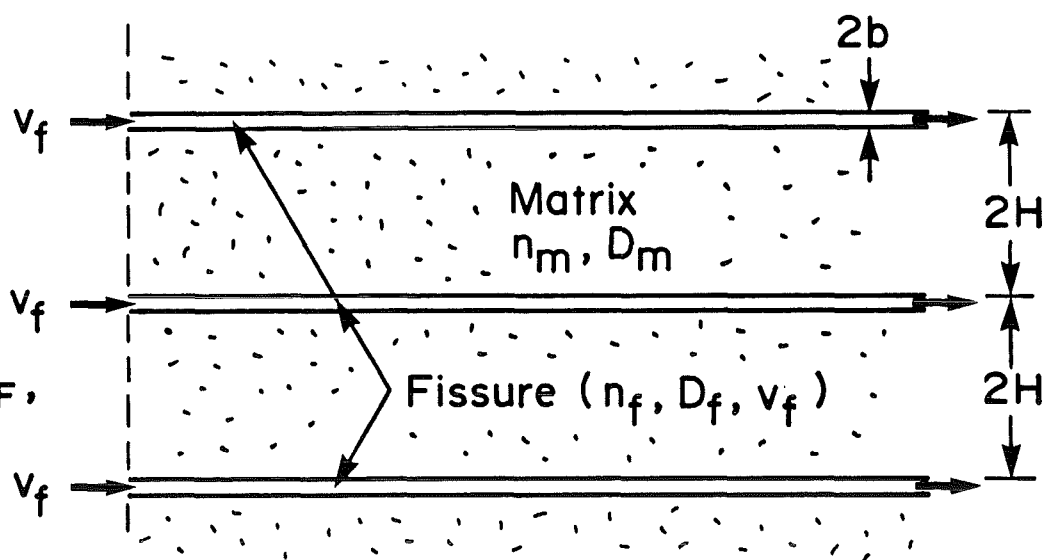


FIGURE 1 PROBLEM IDEALIZATION: PARALLEL CRACKS

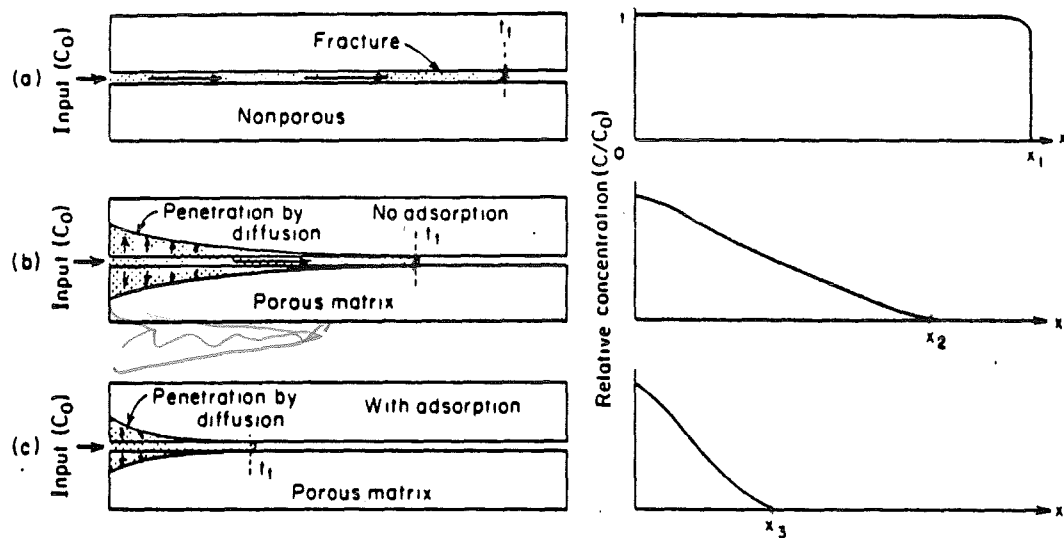
fracture fluid itself then for typical fracture systems where the magnitude of the average groundwater velocity is many orders of magnitude larger than the diffusion coefficient, diffusion would slightly increase the advance of the front (i.e. the distance x_1) and slightly reduce the concentrations behind the front (i.e. rounding off the shape of the contaminant front as indicated in Fig. 2a), however the concentrations at any distance x less than $x_1 = t_1 \cdot v_f$ would be similar to that in the source and should be readily detected if the source concentration c_0 is itself readily detected.

Suppose that the matrix material on either side of the fracture is porous. Once contaminant enters the fracture, the difference in concentration between that in the fracture and the background values in the rock matrix will cause contaminant to move from the fracture into the porous matrix of the rock by a process of molecular diffusion. This movement of contaminant into the rock matrix involves a reduction in the mass of contaminant within the fracture and, since the volume of fluid in the fracture is essentially unaltered, there is a reduction in contaminant concentration in the fracture. Thus, as the contaminant moves along the fracture, more and more of it is removed and the concentration decreases until after some time t_1 , the concentration in the fracture is negligible at some distance x_2 (see Fig. 2b) where x_2 may be substantially less than the distance $x_1 = t_1 \cdot v_f$.

So long as the concentration of contaminant in the leachate is higher than in the fractures, the contaminant concentrations will be greatest at the entrance of the fracture and will decrease both with distance along the fracture and with distance into the matrix of the rock. Contaminant will diffuse into the rock matrix from each fracture and eventually at some point along the fracture, contaminant diffusing from one fracture will meet the contaminant coming from an adjacent fracture. The concentration will then rise within the pore fluid until it

FIGURE 2 (FROM THCL, Site F Report, Vol. 1, Fig. 10, p. 88 -
Original Source - Freeze and Cherry, 1979, p. 411).

ILLUSTRATION OF THE EFFECT OF MATRIX DIFFUSION
(Cherry, 1979)



Effect of diffusion on contaminant migration in porous fractured medium. (a) Unidirectional hydraulic transport in a fracture in a nonporous medium; (b) unidirectional hydraulic transport with migration into matrix as a result of molecular diffusion; (c) unidirectional hydraulic transport with molecular diffusion and adsorption (profiles of relative concentration of reactive contaminant within fracture shown at time t_1).

approaches the value at the face of the fracture. Once this occurs, no more contaminant can move into the rock at that point and the contaminant front will advance to provide additional rock to take up the additional mass of contaminant entering the fractures. Thus the rate of advance of the contaminant front will be related to volume of pore fluid in the matrix, and the time required for contaminant from different fractures to meet. This in turn is related to the matrix diffusion coefficient, D_m , the matrix porosity, n_m , and the spacing of the fractures, $2H$.

When the fracture spacing is close, the contaminant can quickly diffuse into most of the rock mass and hence the apparent rate of movement of the contaminant front (measured in terms of the position where $c/c_0 = 0.1$) will be smaller than that obtained for wider fracture spacings. However, this also means that the pore fluid will reach its maximum concentration more quickly and so the peak concentration reached at a point will be higher than if the spacings were larger.

It may be expected that matrix diffusion will give rise to concentration profiles which have a shape similar to that which would be obtained for an analysis which neglects matrix diffusion but considers some artificial dispersion. Closely spaced fractures will manifest a response similar to that with a low dispersion coefficient; increasing the fracture spacing (for the same total flow) will have an effect similar to increasing the dispersion coefficient in conventional advective-dispersive analyses. It should, however, be emphasized that the mechanisms involved in matrix diffusion are quite different from those involved in mechanical dispersion.

Gillham and Cherry (1982) have reviewed a number of cases where matrix diffusion has been shown to decrease concentrations of species moving along fractures or fissures. For example, Foster (1975) showed that the diffusion of tritium from flowing groundwater in the fractures of porous chalk (i.e. into

WSS
 the pore water of the porous rock matrix) could account for a rapid decrease in tritium concentration in the fractures. Similarly, Gillham and Cherry reported that Day (1977) used this "diffusion to the matrix" concept to account for a rapid decline in tritium concentration with depth in a fractured clay in the Winnipeg area. Finally, Grisak et al. (1980) have also demonstrated that the "diffusion to the matrix" model can give relatively good agreement between theoretical simulations and laboratory experiments in which a tracer solution was passed through a large column of fractured, clayey glacial till. From these studies, Gillham and Cherry concluded that

molecular diffusion can exert a major influence on the rates and patterns of migration of contaminants in fractured argillaceous deposits. Diffusive loss of contaminants from paths of active flow in the fractures to the matrix can be a dominant mechanism of attenuation.

Just as matrix diffusion can result in significant attenuation of conservative contaminant species, it can also result in significant attenuation of reactive contaminants. However, in the case of reactive contaminants, sorption processes will result in even greater attenuation as indicated in Fig. 2c. Some contaminants may be removed from free solution by cation exchange either at the surface of the fracture or within the matrix of the rock. Organics may also be removed by preferential partitioning of contaminants on organic matter (THCL, Vol. 1, pp. 82-83). In principle, partitioning could occur both at the face of the fracture and within the fracture, however the effects of high velocity along the fracture upon the partitioning at the face are uncertain and it would be prudent (and conservative) to restrict consideration to partitioning that occurs within the matrix of the rock.

Models such as that shown in Fig. 1 assume that fractures are continuous within their own plane. It may be anticipated that the real situation is more

complicated and will involve dead end fractures, constrictions (i.e. narrowing followed by widening of the fracture) and a tortuous flow path (eg. flow along a near horizontal fracture then up (or down) a vertical fracture and then along another near horizontal fracture). The net effect of this can be expected to be manifest as a dispersion or spreading of contaminants in excess of that calculated on the basis of advective-diffusive transport along the parallel fractures.

There is no easy way to establish this dispersion coefficient without field data. Similarly, there is uncertainty concerning the value of the groundwater velocity and fracture geometry. Under these circumstances, it is not possible to make precise predictions of contaminant migration no matter how sophisticated the theoretical model used. Fortunately, however, the Burlington and Bayview Park landfills provide an indication of the movement (or lack of movement) of contaminant in a hydrogeologic environment which is similar to that beneath the proposed Halton landfill at Site F. In this situation, theoretical models can be particularly useful for examining the implications of different combinations of parameters and by testing these cases against the known limited extent of migration from the existing landfills, it is possible to form an engineering opinion as to the likely performance of the proposed Halton landfill if situated on Site F.

From the standpoint of modelling contaminant migration, the proposed Site D is at least as complex as Site F and there is uncertainty regarding the value of relevant parameters at both sites. In both cases, modelling using reasonable combinations of parameters will provide an indication of the potential for contamination of the groundwater. At Site F there is the unique advantage of having the documented performance of the two existing landfills which can be used to calibrate the choice of parameters.

3.0 MASS OF CONTAMINANT

THCL performed analyses for the case where (i) the chloride concentration in the landfill remained constant at all times; and (ii) the chloride concentration decreased with time according to the projected curve as shown in Fig. 3. The assumption of a constant source concentration for all time is equivalent to assuming an infinite mass of contaminant is available for transport into the soil. This is clearly not the case in a municipal landfill which has a finite capacity and life. In this regard, THCL commented that (Vol. 1, p. 115):

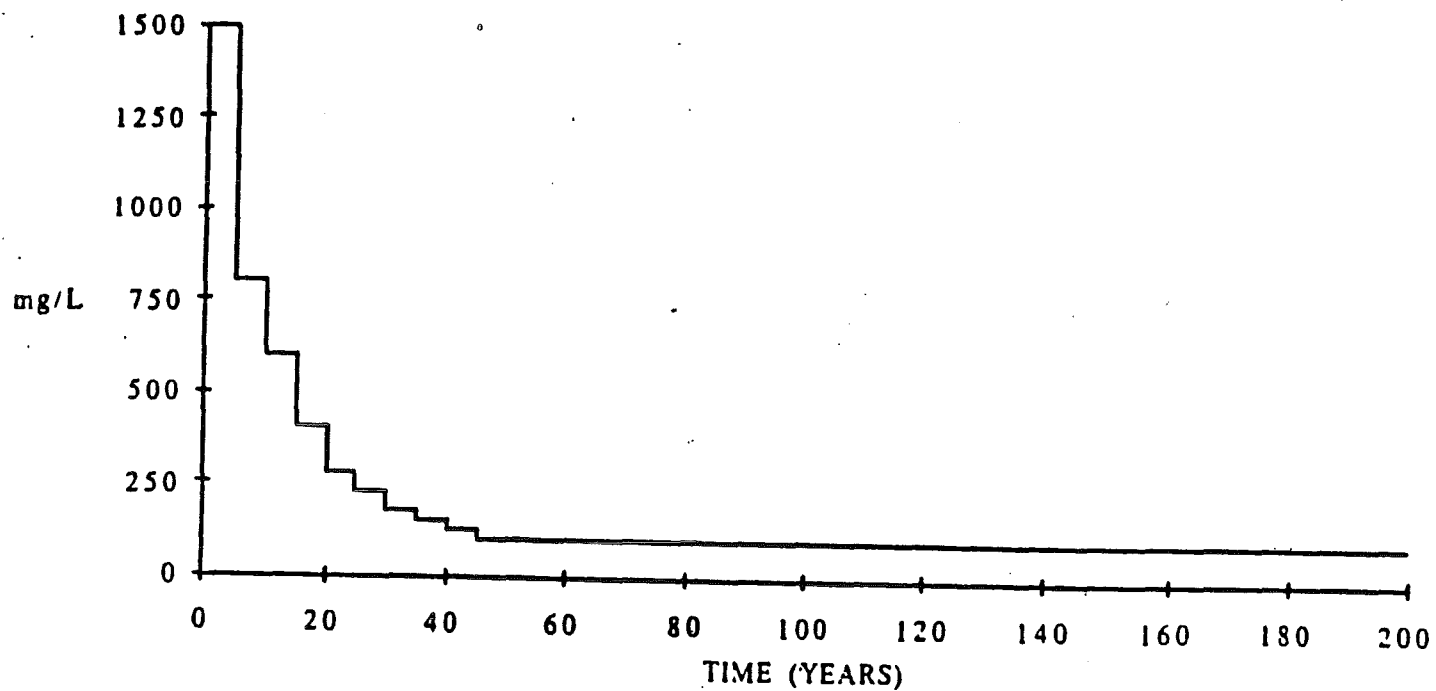
Based on observations at landfill sites and in laboratory leaching experiments, it is inappropriate to assume that contaminant inputs from landfills will remain constant for a period of 200 years. It is generally regarded that contaminant concentrations in municipal landfill leachate will be initially high during operation and shortly following closure, but will decline as more readily leached compounds are removed.

The projected variation in chloride concentration with time adopted by THCL was directly input into their model using superposition of concentrations. This approach can only be adopted if the variation in concentration in the leachate is known. In general, the variation in concentration with time will be a function of the mass of contaminant in the system, the infiltration through the cover, the proportion of leachate collected, and the proportion of leachate passing into the soil. An alternative approach, which considers these factors, has been proposed by Rowe and Booker (see Appendix A) in terms of an "equivalent height of leachate per fracture" H_{LF} and is based on consideration of conservation of mass.

To illustrate the concept, suppose that M_{TC} is the total mass of a particular contaminant species of interest within the landfill. Suppose also that the volume of infiltration into the site, per unit time, is denoted by q_i and the

FIGURE 3 (FROM THCL, Site F Report, Vol. 1, Figure 17, p. 117)

PROJECTED DECLINE IN CHLORIDE
CONCENTRATIONS IN LANDFILL LEACHATE



volume of leachate passing into the underlying rock, per unit time, is q_a (the difference $q_i - q_a$ corresponds to the volume of leachate collected by the collection system), then the proportion of the mass of contaminant which is available for transport along the fractures in the rock is given by

$$M_o = M_{TC} \cdot q_a / q_i \quad (2)$$

Dividing this by the maximum concentration of the contaminant species gives the equivalent volume of leachate V_o , i.e.,

$$V_o = M_o / c_o \quad (3)$$

This volume corresponds to the volume of leachate which would be required if all the mass M_o was initially in solution at the maximum concentration. (It should be noted that this volume of leachate need not correspond to the actual volume of leachate; rather, it is a mathematical convenience based on the conservative assumption that the available mass of contaminant can be quickly leached from the solid waste.) It is also convenient to express the volume V_o in terms of an "equivalent height of leachate per fracture" H_{LF} , where H_{LF} is defined as the volume of leachate divided by the net cross-sectional area through which the contaminant may pass. Assuming that the gross area perpendicular to contaminant flow is A_f (typically the width of the landfill, W , multiplied by the thickness of the fractured rock, T) and the bulk porosity is n_b ($n_b = 2b/2H$), then the net area of fracture through which the contaminant may pass is $A_f \cdot n_b$ and hence

$$H_{LF} = V_o / (A_f n_b) \quad (4a)$$

or

$$H_{LF} = \left(\frac{M_{TC}}{C_o A_f} \right) \cdot \left(\frac{q_a}{q_i} \right) \cdot \left(\frac{1}{n_b} \right) \quad (4b)$$

This representation can be thought of as a column of leachate of width $2b$ which extends back a distance H_{LF} from the face of the fracture.

As indicated in my report on Site D, based on THCL's projected leachate generation of 400 L/min at Site D and the projected concentration up to 45 years (see Fig. 3), the total mass of chloride, M_{TC} , is approximately 4400 tonnes. This is the value which was adopted in the UW0 study of Site D. Since the same waste is to be landfilled at either Site D or F, this mass should apply to either site. Consideration of the entire projected concentration profile (up to 200 years and including THCL's assumption that the concentration remains constant at 100 mg/L for time greater than 45 years) gives a mass of about 7800 tonnes and, to be conservative, this is the value that will be used in this study of the proposed Halton Landfill at Site F.

Assuming that the landfill can be approximated by a rectangular area of length L (parallel to the direction of contaminant transport) and width W (perpendicular to the direction of transport), and assuming that contaminant flows through a thickness of fractured rock denoted by T , then H_{LF} in Equation 4b is equal to

$$H_{LF} = \left(\frac{M_{TC}}{C_o TW} \right) \left(\frac{q_a}{q_i} \right) \left(\frac{1}{n_b} \right) \quad (5)$$

Since contaminant must first move down into the rock beneath the landfill prior to migrating laterally outside the landfill, it may be expected that some attenuation will occur due to diffusion of contaminant into the matrix of this rock beneath the landfill. For fracture spacings between 5 and 30 cm the time required to reach 90% of the equilibrium for contaminant transport into the rock

matrix (beneath the landfill) is between 1 and 10 years. Since this time is small compared with the times required for significant contaminant movement off-site, it is possible to reduce the concentration of contaminant to reflect this attenuation while increasing the equivalent height of leachate (to maintain constant mass). Thus

$$c_o^* = \frac{M_{TC}}{\left(\frac{M_{TC}}{c_o} + LW T n_m\right)} \quad (6a)$$

$$H_{LF}^* = \left(\frac{M_{TC}}{c_o TW} + L n_m\right) \left(\frac{q_a}{q_i}\right) \left(\frac{1}{n_b}\right) \quad (6b)$$

If one assumes that the total mass of a given contaminant within a landfill is proportional to the maximum concentration of that contaminant, then the equivalent height of leachate per fracture will be the same for all contaminant species.

4.0 PROBLEM IDEALIZATION

Examination of the available data indicates that the Queenston shale in the environs of the proposed Halton landfill on Site F is fractured. The fracture spacing varies, but is typically between 5 cm and 30 cm. Based on the available information, it would appear that leachate which is not collected by the leachate collection system will move down into the fractured rock beneath the landfill and then move laterally through the fractured rock in a southeasterly direction.

For the purpose of assessing the potential for contaminant migration it will be assumed that the contaminant moves down into the rock mass beneath the site and that this rock mass acts as the contaminant source for subsequent lateral

contaminant migration along a series of parallel fractures with a spacing of $2H$ and a fracture width $2b$ as shown in Fig. 1. The model used to examine migration along these fractures is an extension of a model proposed by Sudicky and Frind (1982) in that it considers one-dimensional advective-dispersive transport along a series of parallel fractures and one-dimensional diffusive transport into the matrix of the shale between the fractures. "Sorption" of non-conservative contaminants within the rock matrix was also modelled in terms of a linear distribution or partitioning coefficient K_d . The primary difference between the model adopted in the UWO study (see Appendix A) and the Sudicky and Frind model is that Sudicky and Frind assumed a constant source concentration c_0 (i.e. infinite mass of contaminant) whereas in the present formulation the concentration of contaminant in the source is controlled by considerations of conservation of mass and the initial mass of contaminant available for transport into the rock (expressed in terms of the "equivalent height of leachate per fracture" H_{LF} as previously discussed).

The following discussion relates to water movement in the deep groundwater system. The parameters controlling the movement of contaminant in the system of parallel fractures considered in the model are:

- (1) the Darcy velocity, v_a , which represents the average lateral flow (per unit area) within a deep groundwater system. This velocity will depend on the hydraulic conductivity of the rock and the lateral hydraulic gradients. If contaminant is considered to be moving parallel to the axis of the landfill (assuming a landfill of length L in this direction) through a fractured rock mass of thickness T , then continuity of flow implies that

$$v_a > q_a L / T \quad (7)$$

where q_a is the flow (per unit area) of leachate into the rock from the landfill

- (2) the spacing of the fractures, $2H$
- (3) the width of the fractures, $2b$
- (4) the porosity of the rock with respect to fracture flow n_b . Note that this is dependent on the spacing and width of fractures and is given by $n_b = (2b)/(2H)$
- (5) the average linearized groundwater velocity v_f within the fractures. Note that this quantity is in fact derived from v_a , H and b and is not an independent quantity, viz.

$$v_f = v_a / n_b \quad (8)$$

- (6) the coefficient of hydrodynamic dispersion D_f within the fracture where

$$D_f = D_{of} + \alpha v_f \quad (9)$$

math. law

and D_{of} is the diffusion coefficient for contaminant in the fracture (taken to be the diffusion coefficient for the contaminant of interest in aqueous solution); and

chemistry - lab value

α is the dispersivity

- (7) the diffusion coefficient D_m for diffusion of the contaminant of interest into the rock matrix

- (8) the porosity n_m of the rock matrix *for all intact rock.*

- (9) the distribution or partitioning coefficient for "sorption" of the contaminant of interest as it moves through the rock matrix

- (10) the reference concentration, c_o , or c_o^* of contaminant *initial concentration* in the source where c_o is the maximum value expected in the landfill itself and c_o^* is an adjusted

concentration for the rock beneath the landfill and is given by Eq. 6a

(11) the equivalent height of leachate per fracture as determined from Eq. 6b.

5.0 SELECTION OF PARAMETERS

Analyses were performed to assess

- (i) the migration of contaminants from the existing Burlington landfill 15 years after migration commenced based on a number of assumptions concerning the average flow per unit area (Darcy velocity v_a), the fracture geometry ($2H$, $2b$, n_b), and the dispersivity (α). The results from similar calculations for periods of 10 and 25 years are given in Appendix B; and
- (ii) the migration and attenuation of contaminants from the proposed Halton landfill at Site F using parameters judged to be reasonable based on the available data and the examination of the impact of these parameters on the calculated and observed migration from the Burlington landfill (i.e. (i) above).

The selection of the parameters used in these two series of analyses is described below.

5.1 Assumed Landfill Dimensions

Based on Drg. 5r given by THCL in Vol. 3 of their Site F report, the Proposed Halton Landfill was represented by a rectangular area of length $L = 1250$ m and width $W = 400$ m giving a total plan area of 50 ha.

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

$$\frac{800 \times 250}{10000}$$

20 ha

on Drg. 1, GLAL, 1986).

5.2 Lateral Flow (Darcy Velocity) in the Deep Groundwater System

The lateral Darcy velocity (or flow) in the rock can be estimated from field values of the hydraulic gradient and hydraulic conductivity or from consideration of continuity of flow. Based on available data and advice from Geocon, it would appear that a lateral Darcy velocity of $v_a = 0.05$ m/a might be expected to be representative of flow conditions in the deep groundwater system in the environs of Site F. For a fracture porosity $n_b = 0.001$, this Darcy velocity corresponds to a lateral fracture velocity (average linearized groundwater velocity) $v_f = 50$ m/a. This appears to be consistent with values reported by THCL (Vol. 1, p. 6) to range between 6 m/a and 130 m/a.

The lateral Darcy velocity of $v_a = 0.05$ m/a could also be thought of in terms of an average hydraulic conductivity of 4.5×10^{-6} cm/s and a lateral hydraulic gradient of 0.035. It is noted that, excluding the very shallow test at about 2 m, this hydraulic conductivity is higher than that measured by Hewetson (1985, p. 27) and is similar to the value estimated by GLAL (1984, p. 0-30).

For the purposes of considering a "worst case" scenario in the event that the leachate collection tiles all failed it was very conservatively assumed that leachate mounding would develop to the top of the waste mound. This assumption is more severe and conservative than that made in the study of Site D even though one would expect the height of mounding to be less at Site F than at Site D due to the sloping nature of the site. Based on this very conservative assumption, the lateral flow per unit area (Darcy velocity) in the deep rock system was estimated to increase from .05 m/a to 0.125 m/a. This corresponds to an increase in the lateral gradient by a factor of 2.5 due to the mounding and is based on the

assumption that the head 300 m downgradient would not change due to mounding. If the head at this point did increase then the flow would be even smaller than indicated above. For a fracture porosity of 0.001, this corresponds to a fracture velocity (average linearized groundwater velocity) of 125 m/a.

The lateral Darcy velocities of 0.05 m/a and 0.125 m/a quoted above can be related to the infiltration over the area of the proposed landfill as indicated by Equation 7. For example, suppose that the recharge area corresponds to the long axis of the proposed landfill (i.e. $L = 1250$ m) and that the majority of the lateral deep groundwater flow will occur through a 25 m thick layer of fractured rock (i.e. $T = 25$ m). Adopting these values of L and T , then Equation 7 indicates that the Darcy velocities of 0.05 m/a and 0.125 m/a correspond to an average infiltration into the deep groundwater system of 1 mm/a and 2.5 mm/a respectively. If one were to assume that the flow extended to 50 m depth, the corresponding infiltration values would increase to 2 mm/a and 5 mm/a. It should be noted that the calculations reported for Site F in the following section were performed for $T = 25$ m however the results are very insensitive to the thickness T used in the calculations and the conclusions reached would be the same irrespective of the actual thickness used in the analysis (for any reasonable value for T).

THCL's computer model of current conditions (THCL, Vol. 1, p. 74) gives an infiltration q_a into the rock of approximately 14 mm/a. For flow through a 25 m thick section of rock, this implies that the Darcy velocity in the rock would be greater than, or equal to, 0.7 m/a. If one considers THCL's projected inflow of 22 mm/a after construction of the Halton landfill then this corresponds to a Darcy velocity of 1.1 m/a. Assuming a porosity of 0.001, these Darcy velocities correspond to fracture velocities of 700 m/a and 1100 m/a respectively!

To provide some appreciation of the implication of adopting an average Darcy

velocity of 0.05 m/a (based on expected conditions) compared to a Darcy velocity of 0.7 m/a (based on THCL's indicated infiltration), analyses were performed for the existing Burlington Landfill for these two values of Darcy velocity.

For the Burlington Landfill, the Darcy velocity of 0.05 m/a passing through a 25 m thick layer of rock corresponds to an average recharge over the entire site of about 1 mm/a (i.e. similar to Site F). If one were to make the extreme assumption that there is no recharge under the portion of the site where there is a leachate underdrain system, then the Darcy velocity of 0.05 m/a flowing through 25 m of rock would correspond to an average infiltration into the deep groundwater system of about 1.5 m/a over the southern portion of the site where there is no leachate underdrain system. As noted earlier, if one were to consider flow through, say, 50 m of rock at the same Darcy velocity, this would correspond to infiltration double that indicated above but would not have any significant effect on the results or conclusions reached in the following section.

For the Burlington Landfill, the Darcy velocity of 0.7 m/a passing through 25 m of fractured rock corresponds to an average groundwater recharge over the entire area of the existing Burlington Landfill of about 14 mm/a, which is infiltration predicted by THCL's finite element model for Site F prior to construction of a landfill (THCL, Vol. 1, p. 74). Alternatively, if one again makes the assumption that there is no infiltration into the deep groundwater system at the north end of the site (where there is a leachate collection system), then the Darcy velocity of 0.7 m/a flowing through 25 m of fractured rock corresponds to an infiltration over the southern portion of the site equal to about 22 mm/a (i.e. this approximately corresponds to the infiltration predicted by THCL's finite element model after construction of the Site F landfill).

It should be emphasized that the critical parameter in analyses of contaminant migration in fractured systems is the average lateral flow (i.e. Darcy

velocity). Varying the thickness, T , of the flow zone and the infiltration q_i does not significantly affect the results of the analysis provided that the Darcy velocity (as given by Eq. 7) remains the same.

5.3 Fracture Spacing and Aperture Size

Based on the THCL core logs and advice from Geocon, the typical fracture spacing was taken to be between 5 cm and 30 cm.

Test pits by GLAL (1979, p. 12) indicated that:

The fractures are predominantly tight, less than 0.08 mm (80 μ m) in size, although some fractures up to 0.40 mm (400 μ m) were noted.

It should be noted that these fractures were near the surface and hence due to stress relief may have opening sizes greater than at depth. Based on available data, 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 μ m and typically less than 500 μ m (0.5 mm) with a probable range between 10 μ m and 50 μ m (Hoek and Bray, 1981). For a spacing of 5 cm this corresponds to a range in porosities n_D of between 0.0001 and 0.01.

5.4 Diffusion Coefficients, Retardation Factors and Matrix Porosity

To provide some indication of the level of attenuation which might be expected, consideration was given to the migration of both a conservative contaminant species (chloride) and a volatile priority pollutant (benzene).

Chloride was selected because it is generally regarded as a primary indicator of the presence of a contaminant plume. The diffusion coefficient for chloride migration along a fracture at an assumed ground temperature of about 7°C

was taken to be equal to $0.04 \text{ m}^2/\text{a}$ ($12.7 \times 10^{-6} \text{ cm}^2/\text{s}$) (i.e. the diffusion coefficient in free solution, based on Quigley et al., 1987). To determine the diffusion coefficient for chloride migration in the matrix of the shale, three diffusion tests were performed on samples of Queenston shale taken from a depth of approximately 11.4-11.8 m at Site F. The diffusion coefficient determined from these tests ranged from 1.4×10^{-6} to $1.6 \times 10^{-6} \text{ cm}^2/\text{s}$ with an average value of $1.5 \times 10^{-6} \text{ cm}^2/\text{s}$ ($0.0047 \text{ m}^2/\text{a}$). Attenuation due to matrix diffusion will be greatest when the matrix diffusion coefficient is high and so for this type of modelling it is conservative to adopt a low value of the diffusion coefficient. To ensure that the calculations performed were conservative, the matrix diffusion coefficient for chloride was reduced by a factor of 2.5. Thus the value used in this study was assumed to be $0.0019 \text{ m}^2/\text{a}$ (i.e. $6 \times 10^{-7} \text{ cm}^2/\text{s}$) which corresponds to a tortuosity of 0.047. This value is very close to the value of $5 \times 10^{-7} \text{ cm}^2/\text{s}$ adopted by THCL.

Based on the measured water content of the shale, the matrix porosity n_m of the shale was determined to be between 0.10 and 0.11. A value of 0.1 was adopted by THCL (Vol. 1, p. 97). To be conservative, the lower value of $n_m = 0.08$ was adopted in the UWO study.

Benzene was considered because it is a relatively mobile priority pollutant having a diffusion coefficient in free solution (at 10°C) of approximately $0.024 \text{ m}^2/\text{a}$ ($7.5 \times 10^{-6} \text{ cm}^2/\text{s}$; Reid et al., 1977). The diffusion coefficient in the shale matrix was obtained by assuming that the tortuosity factor for benzene would be similar to that of chloride (viz. 0.047) giving $D_m = 0.0011 \text{ m}^2/\text{a}$ ($3.6 \times 10^{-7} \text{ cm}^2/\text{s}$). This value is also considered to be quite conservative because the tortuosity has been factored down as discussed above.

Based on empirical correlations (Schwarzenbach and Westall, 1981) between partitioning coefficient K_d , the fraction of organic carbon ($f_{oc} = 0.18\%$,

THCL, Vol. 1, p. 100) and the octanol/water partitioning coefficient (Chiou and Schmedding, 1982), the partitioning coefficient was estimated to be $0.19 \text{ cm}^3/\text{g}$. Based on this value of K_d , densities ρ_m in the range $2.4\text{-}2.5 \text{ g/cm}^3$ and matrix porosities n_m in the range $0.08\text{-}0.1$, the retardation factor $R_m = (1 + \rho_m K_d / n_m)$ ranges from 5.5 to 7. To be conservative, a value of $R_m = 5.5$ was used for benzene in the analyses reported herein.

5.5 Dispersivity

The dispersivity α in the fractures can not be measured directly for the fractured system at Site F. To illustrate the effect of this parameter, analyses were performed for a number of dispersivities as discussed in the following sections.

5.6 Initial Concentration and Mass of Contaminant

To be consistent with the studies conducted by THCL and with the UWO study of Site D, a reference concentration for chloride was taken to be 1500 mg/L . The reference concentration for benzene was taken to be $150 \text{ }\mu\text{g/L}$ based on the data given in Table 11, THCL (Vol. 1, p. 92) for the Burlington Landfill.

For the analysis of the existing Burlington Landfill, a mass of chloride of 2200 tonnes and a mass of benzene of 0.22 tonnes (i.e. 250 L or 55 gallons of pure benzene) were assumed and hence from Equation 6a the value of c_0^* is 1180 mg/L for chloride and 118 mg/L for benzene. From Eq. 6b, the corresponding value of H_{LF}^* is

$$H_{LF}^* = \frac{360(q_a/q_i)}{n_b}$$

where these calculations and the mass of contaminant is only based on the portion

of the Burlington Landfill which does not have a leachate underdrain system. For this case it was assumed that all the mass of contaminant is available for transport into the rock. 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}) would not affect the conclusions reached in the following section.

For the analysis of the proposed Halton landfill at Site F, the masses of chloride and benzene were taken to be 7800 tonnes and 0.78 tonnes respectively, corresponding to the most conservative interpretation of the concentration variation proposed by THCL and shown in Fig. 3. For the case of benzene, this mass corresponds to a volume of approximately 880 L (almost 200 gallons) of pure benzene. Allowing for a reduction in concentration as the contaminant moves down into the rock beneath the landfill gives (from Eq. 6a) c_o^* of 1260 mg/L for chloride and 126 $\mu\text{g/L}$ for benzene. Using Eq. 6b, the equivalent height of leachate per fracture obtained for these masses of contaminant is

$$H_{LF}^* = \frac{620 q_a / q_i}{n_b}$$

To be consistent with the analyses performed for Site D, q_i was again taken to be 0.3 m based on THCL's projected infiltration rate of 5.8 L/min/ha (THCL, Vol. 3, p. 69).

6.0 ANALYSIS OF THE EXISTING BURLINGTON LANDFILL

6.1 Migration of Benzene and Similar Organics

The benzene plume from the existing Burlington landfill was calculated for 11 different combinations of parameters as summarized on Fig. 4 and in Table 1. Cases 1-6 are based on the assumption that the lateral flow per unit area (i.e.

TABLE 1 CASES CONSIDERED

Case	Contami- nant	c_o (mg/L)	c_o^* (mg/L)	q_a (m/a)	v_a (m/a)	2H (m)	2b (μ m)	n_b	v_f (m/a)	α (m)	D_f^1 (m^2/a)	D_m (m^2/a)	R	Landfill
1	"Benzene"	0.15	0.118	0.014	0.7	1.2	30	0.000025	28000	0	0.024	0.0011	5.5	Burlington
2	"	"	"	"	"	0.3	30	0.0001	7000	"	"	"	"	"
3	"	"	"	"	"	0.3	300	0.001	700	"	"	"	"	"
4	"	"	"	"	"	0.05	5	0.0001	7000	"	"	"	"	"
5	"	"	"	"	"	0.05	50	0.001	700	"	"	"	"	"
6	"	"	"	"	"	0.05	500	0.01	70	"	"	"	"	"
7	"	"	"	0.001	0.05	0.3	300	0.001	50	"	"	"	"	"
8	"	"	"	"	"	0.05	50	"	"	"	"	"	"	"
9	"	"	"	"	"	0.3	300	"	"	2.5	125	"	"	"
10	"Chloride"	1500	1180	0.014	0.7	0.3	300	0.001	700	0	0.04	0.0019	1.0	Burlington
11	"	"	"	"	"	0.05	50	"	700	0	0.04	"	"	"
12	"	"	"	0.001	0.05	0.3	300	"	50	0	0.04	"	"	"
13	"	"	"	"	"	0.3	300	"	"	2.5	125	"	"	"
14	"	"	"	"	"	0.05	50	"	"	0	0.04	"	"	"
15	"	"	"	"	"	0.05	50	"	"	2.5	125	"	"	"
16	"	"	1260	"	"	0.3	300	"	"	30	1500	"	"	Halton
17	"	"	"	"	"	0.05	50	"	"	30	1500	"	"	"
18	"	"	"	"	"	0.05	50	"	"	80	4000	"	"	"
19	"Benzene"	0.15	0.126	0.001	"	0.3	"	"	"	30	1500	0.0011	5.5	"
20	"	"	"	"	"	0.05	"	"	"	30	1500	"	"	"
21	"	"	"	"	"	"	"	"	"	80	4000	"	"	"
22	Organic B	2.0	1.68	"	"	"	"	"	"	80	4000	0.00095	1.02	"
23	Organic C	3.0	2.52	"	"	"	"	"	"	80	4000	0.00095	2.0	"
24	"Chlorine"	1500	1260	0.0025	0.125	0.05	50	0.001	125	30	3750	0.0019	1.0	Halton
25	"Benzene"	0.15	0.126	"	"	"	"	"	"	30	3750	0.0011	5.5	"
26	"Benzene"	0.15	0.118	"	"	0.3	300	"	"	2.5	312.5	"	5.5	Burlington
27	"	"	"	"	"	0.05	50	"	"	2.5	312.5	"	"	"
28	"Chlorine"	1500	1180	"	"	0.3	300	"	"	2.5	312.5	0.0019	1.0	"
29	"	"	"	"	"	0.05	50	"	"	2.5	312.5	"	"	"

Note: ¹ $D_f = D_{of} + \alpha v_f$ ² $n_m = 0.08$ for all cases

the Darcy velocity, v_a , of the groundwater) is 0.7 m/a. Cases 7-9 were obtained from a lateral flow per unit area (v_a) of 0.05 m/a. Cases 26-27 were obtained for $v_a = 0.125$ m/a.

Case 1 assumes a relatively massive rock system with a fracture spacing of 1.2 m. For this case, contaminant is calculated to exceed $40 \mu\text{g/L}$ up to 300 m southeast of the landfill and should have been readily detectable in boreholes GL 12, 13, 15, 16, 17, 24 and 27. The fact that the Burlington Landfill Plume Delineation Study (GLAL, 1986) has not detected significant quantities of volatile organics (i.e. in excess of $15 \mu\text{g/L}$) even 25 m from the landfill (i.e. at GL 24), would strongly suggest that the parameters used in this analysis do not provide a good representation of the actual field conditions.

Cases 2-8 involve fracture spacing of 5 cm and 30 cm which bracket the typical spacing of fractures reported in the core logs. Assuming that the dispersivity is zero and that the total flow through the porous rock remains constant, cases 2 and 3, and cases 4, 5 and 6 illustrate the effect of changing the assumed fracture aperture size $2b$ (and hence the porosity $n_b = 2b/2H$) over a very wide range for fracture spacings of 30 cm and 5 cm respectively. Inspection of the results shown in Fig. 4 indicates that to all practical purposes, the fracture opening size ($2b$) and the fracture porosity has no direct effect on the concentration plume for the assumed conditions. This is an important finding since it implies that uncertainty as to the precise value of the fracture opening size and the fracture porosity will have negligible effects on the calculated migration of contaminants. ~~What is important, however, is the spacing of fractures and the total volume of fluid flowing through the fractured system (i.e. the Darcy velocity).~~

The influence of the fracture spacing can be appreciated by comparing the results obtained for cases 2 and 3 ($2H = 30$ cm) with those obtained for cases 4,

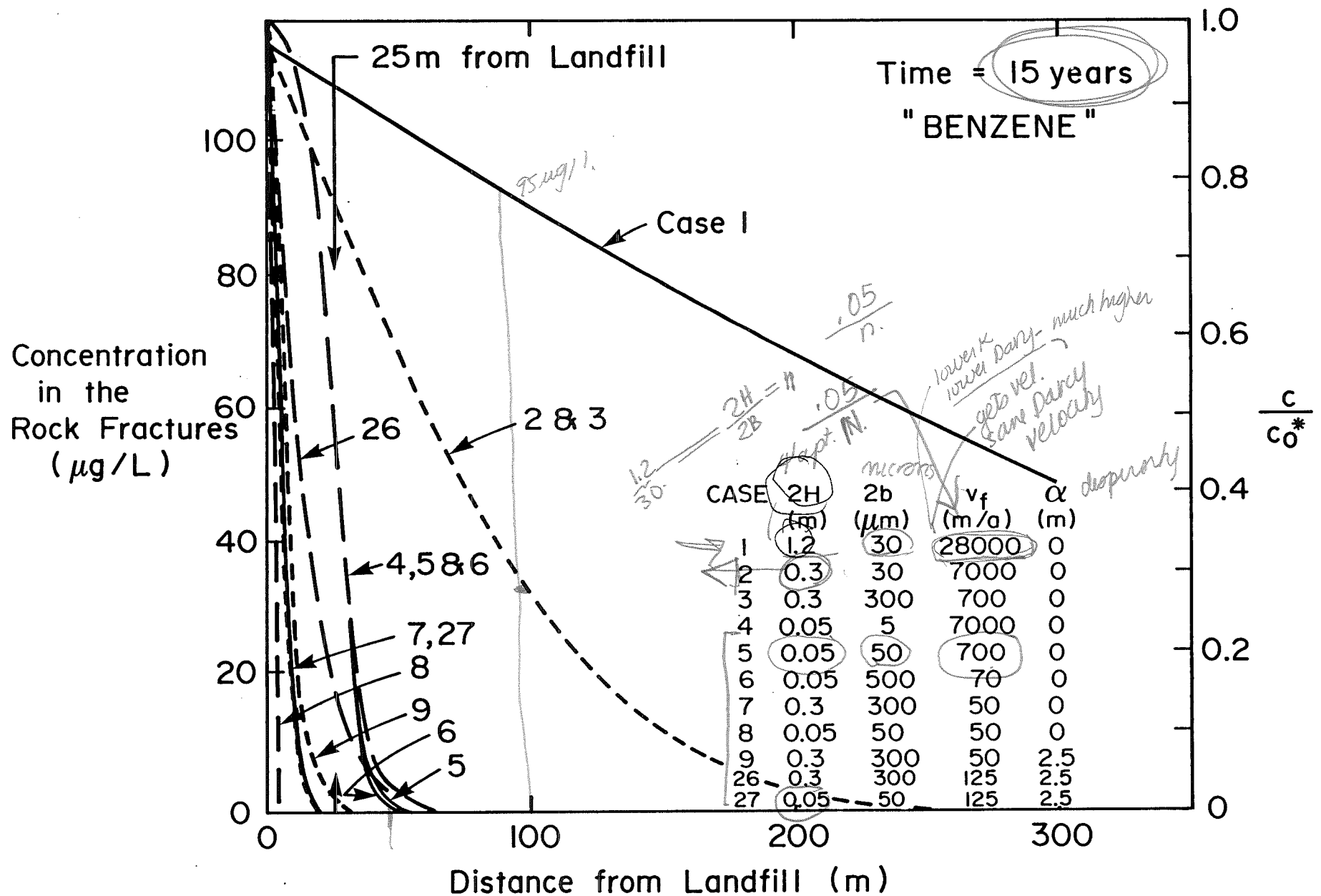


FIGURE 4 CALCULATED MIGRATION OF BENZENE FROM THE BURLINGTON LANDFILL AFTER 15 YEARS

5 and 6 ($2H = 5$ cm). It can be seen that the wider the fracture spacing, the faster the advance of the contaminant plume (other things being equal). This situation arises because the wider spacing between fractures means that it takes longer for contaminant to move into the entire rock matrix and hence the effect of matrix diffusion is less significant for the time scale of interest here. If one considers the spacing of 5-30 cm to bracket the expected fracture spacing of the rock then examination of Fig. 4 would suggest that if there is a groundwater recharge of 14 mm/a from the existing Burlington landfill into the underlying rock, then benzene should be detectable at levels exceeding $10 \mu\text{g/L}$ for distances of between 40 to 160 m from the landfill, assuming no dispersion in the fractures. If dispersion in the fractures was considered, then contaminant should be detected even further from the landfill. If the advective velocity used in this calculation was correct, then monitoring at borehole GL 24 should have detected concentrations of benzene exceeding $50 \mu\text{g/L}$ after 15 year migration. One would expect contaminants such as 1,2 Dichloroethane, 1,1 Dichloroethylene, Methylene chloride, Diethyl ether, Dimethyl sulphide and Methyl ethyl ketone to be even further advanced and to be detectable at borehole 24 at much higher concentrations than benzene.

The influence of the volume of fluid flowing through the fractures is evident from a comparison of the results for cases 3 and 7, or 5 and 8 which show the effect of the difference in lateral flow per unit area (Darcy velocity) between 0.7 m/a and 0.05 m/a . It can be seen that reducing the Darcy velocity substantially increases the level of attenuation (all other factors being equal). Attenuation is, in fact, dependent on the total mass of contaminant being transported along the fracture (and hence the Darcy velocity) rather than the actual speed of the transport (i.e. the average linearized groundwater velocity).

Assuming a dispersivity of zero, it can be seen that for the representative range of fracture spacing and $v_a = 0.05$ m/a, benzene would be expected to be at concentrations less than $1 \mu\text{g/L}$, 25 m from the existing Burlington landfill after 15 years. If one assumes a dispersivity of 2.5 m (i.e. 10% of 25 m travel distance of interest) then for the widest value of the expected fracture spacing ($2H = 30$ cm; case 9 on Fig. 4) benzene would be detected at a concentration of about $2 \mu\text{g/L}$ at points 25 m from the landfill after 15 years migration.

Based on this information, it is considered unlikely that benzene would be detected in any significant quantity at any of the Gartner Lee boreholes south-east of the existing Burlington Landfill after only 15 years migration through the rock if the average flow per unit area (Darcy velocity) is of the order of 0.05 m/a. However, if the average flow were more than an order of magnitude larger (specifically 0.7 m/a) then based on these calculations, substantial quantities of benzene (or similar organic contaminants) should be detected at borehole GL 24 (and similarly at GL 12, if sampling was to be conducted at this borehole).

Analyses of contaminant migration from the portion of the existing Burlington landfill which does not have a leachate underdrain system were also performed adopting a lateral Darcy velocity of 0.125 m/a (cases 26 and 27, Fig. 4).* The results of these analyses indicate that benzene might be detected 25 m from the landfill (eg. at GL 24). The concentration after 15 years migration could be up to 15% of the source concentration if the average fracture spacing was about 30 cm (Case 26). If the fracture spacing was 5 cm (Case 27) then it is unlikely that benzene would be detectable (at the $1 \mu\text{g/L}$ level) 25 m from the landfill after 15 years migration.

* This corresponds to the worst case scenario for Site F involving leachate mounding to the top of the waste.

6.2 Migration of Conservative Contaminants

The foregoing discussion has concerned an EPA priority organic pollutant which would experience attenuation due to both matrix diffusion and "sorption". To be consistent with THCL's attempt to provide an indication of the potential migration of conservative species of contaminant, analyses were also performed for chloride. In the following discussion, chloride is being used as representative of highly mobile non-reactive contaminants and the comments should not be regarded as being directly applicable to chloride. This qualification is necessary because the very poor quality of groundwater in this area involves concentrations of chloride in the rock which greatly exceed values expected in the leachate. Due to diffusion of chloride out of the rock matrix and into the fractures, the chloride concentration in infiltrating leachate or groundwater should actually increase with distance from the landfill rather than be attenuated. Thus, in terms of chloride, the water quality should decrease with distance from the recharge area due to "contamination" from the naturally occurring Queenston shale.

Although the calculations were performed for chloride, the results may be considered to be applicable to any conservative contaminant species with a diffusion coefficient similar to that of chloride. Conservative species with a diffusion coefficient less than that of chloride would migrate farther than chloride in the same time.

Keeping in mind the foregoing caveat regarding the actual applicability of the results to chloride itself, Figs. 5 and 6 show the calculated chloride profiles after 15 years for seven combinations of parameters. Cases 10 and 11 correspond to a Darcy velocity of 0.7 m/a. For the expected range of fracture spacings, one would expect a conservative contaminant like chloride to be detectable

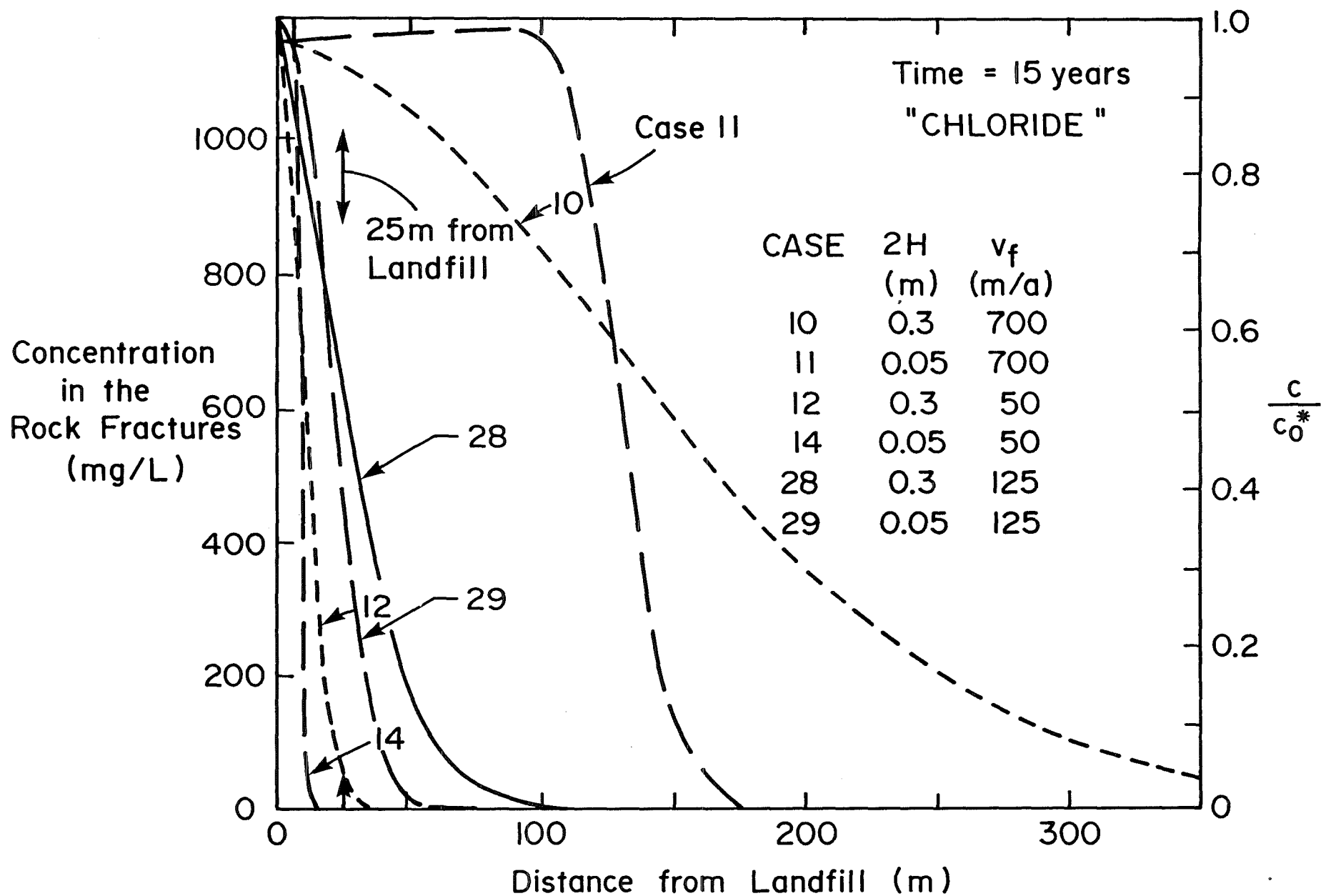


FIGURE 5 CALCULATED MIGRATION OF "CHLORIDE" FROM THE BURLINGTON LANDFILL AFTER 15 YEARS

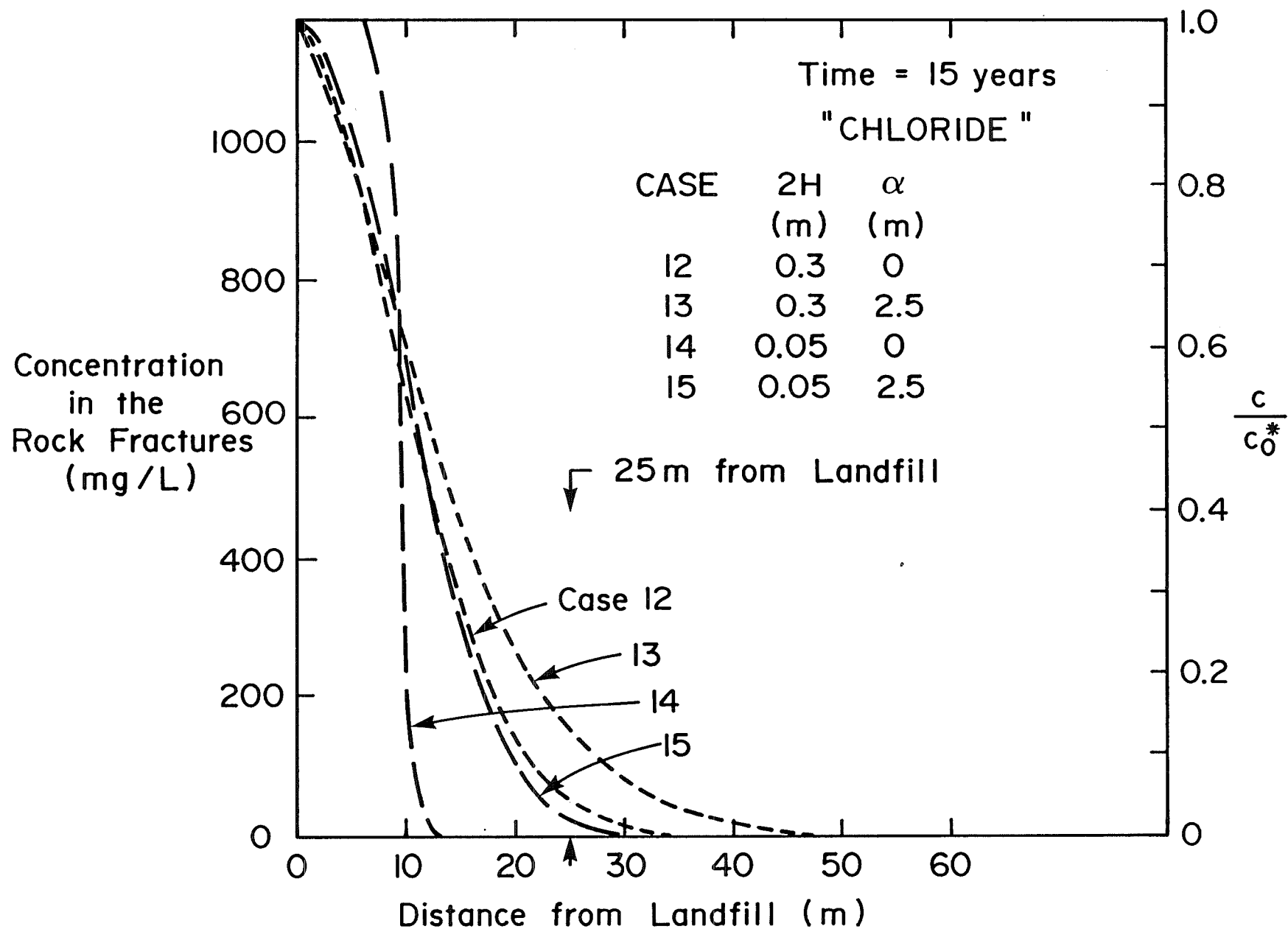


FIGURE 6 CALCULATED MIGRATION OF "CHLORIDE" FROM THE BURLINGTON LANDFILL AFTER 15 YEARS

at a level exceeding 10% of the initial value (c_0) in leachate at a distance of at least 150 m and possibly as much as 275 m from the landfill. Thus, if the Darcy velocity were 0.7 m/a to the southeast of the existing Burlington landfill, then conservative species should be readily detectable at boreholes GL 12, 16 and 24 and probably also at GL 13. The concentration at GL 12 and 24 should exceed 50% of the value in the leachate. ^{don't know} The fact that these high levels of conservative contaminants are not observed in these boreholes is considered to be good evidence that the advective velocity of contaminant transport is much smaller than the 0.7 m/a adopted in these calculations and hence the average recharge of the groundwater is expected to be less than 14 mm/a. In support of this conclusion, it is also noted that for this Darcy velocity, high concentrations of contaminant would be calculated at distances of up to 75 m for any reasonable combination of fracture spacing, fracture aperture, fracture porosity, or dispersivity.

Cases 12-15 correspond to an average flow per unit area (Darcy velocity) of 0.05 m/a (i.e. an average recharge of 1 mm/a). Case 12 indicates that for a fracture spacing of 30 cm, the contaminant plume calculated using a dispersivity of zero would be detectable 25 m from the landfill at a level of about 3% of the source concentration. If the fracture spacing was significantly closer than 30 cm, then no contaminant would be detected. If one considers a dispersivity of 2.5 m (i.e. 10% of the 25 m distance of interest), then for the expected range of fracture spacing, contaminant would be detectable at levels of 3 to 10% of the original source concentration level in the landfill. A similar conclusion could be reached by repeating the analysis for a slightly higher Darcy velocity and no dispersivity, or for a lower Darcy velocity and higher dispersivity, however the net result is that for Darcy velocity of the order of 0.05 m/a in a southeasterly

direction, one would expect that conservative contaminant might be just detectable 25 m from the landfill and not detectable at boreholes which are located further from the landfill.

Analyses performed adopting a lateral Darcy velocity of 0.125 m/a (corresponding to the "worst case" leachate mounding at Site F - Cases 28 and 29) indicate that for these conditions, conservative species would be readily detected 25 m from the landfill (i.e. at GL 12 and 24) and might be detected at low concentrations at GL 16 if the average fracture spacing is approximately 30 cm (Case 28). For a fracture spacing of 5 cm (case 29) conservative contaminants would probably not be detected at GL 16 after 15 years migration.

6.3 Summary

It is apparent from the results presented in Figs. 4-6 that the Darcy velocity of the groundwater is the key parameter controlling the migration of contaminant from any landfill constructed on the fractured shale at or adjacent to the proposed Site F. Making all reasonable allowances for uncertainty regarding fracture spacing, fracture aperture, fracture porosity and dispersivity, it is clear that for an average lateral Darcy velocity of 0.7 m/a (eg. corresponding to an average groundwater recharge of 14 mm/a assuming fractured flow through 25 m thickness of rock), high concentrations of conservative contaminant species and high concentrations of priority pollutant such as benzene should be detectable at a number of boreholes including GL 12 and 24 with the additional possibility that they would be detected in smaller concentrations at boreholes GL 13 and 16. The fact that this has not been observed by the Burlington Landfill Plume Delineation Study (GLAL, 1986, p. 40) provides evidence that the Darcy velocity is far smaller than assumed in these calculations.

The foregoing conclusion is primarily based on the assumption that the flow is in a southeasterly direction, however even in the event that flow was in either a southerly or easterly direction, the calculations indicate that conservative contaminants should be at readily detectable levels at boreholes GL 15 or 32 respectively.

On the other hand, the concentration plume calculated for a Darcy velocity of 0.05 m/a implies that conservative contaminant would be detected 25 m from the landfill (assuming flow in the southeasterly direction) but not at more distant boreholes. Priority pollutants would probably not be detected at any of the boreholes. However, if the average fracture spacing is about 30 cm and if there is also a moderate amount of dispersion then, based on these calculations, a small amount of organics (~3% of c_0) might be detected 25 m from the landfill. The fact that the Burlington Landfill Plume Delineation Study (GLAL, 1986) did not detect benzene at levels exceeding 1 $\mu\text{g/L}$ lends support to the conclusion that the average flow per unit area in the southeasterly direction is about 0.05 m/a and that the average fracture spacing is probably less than 30 cm but more than 5 cm.

7.0 ANALYSIS OF THE PROPOSED HALTON LANDFILL - SITE F

One of the key questions regarding the proposed landfill at Site F is as to what impact the leachate may have on the groundwater in the rock southeast of the landfill. To provide some indication of potential impact, analyses were performed for both a conservative species of contaminant (chloride - cases 16-18 & 24 as indicated in Table 1) and a priority pollutant (benzene - cases 19-21 & 25).

For the purposes of these calculations, the initial (reference)

concentration in the landfill c_0 was assumed to be 1500 mg/L and 150 μ g/L for chloride and benzene respectively. These initial concentrations correspond to values measured in leachate at the existing Burlington landfill (THCL, Vol. 1, p. 92) however the value for benzene is considered to be very conservative since it greatly exceeds that found in other landfills (see THCL, Vol. 1, pp. 91-93; THCL have suggested that the high concentration of volatile organic chemicals in the Burlington landfill is the result of the disposal of some industrial waste). The corresponding volume of pure benzene assumed in the analysis is about 880 L (i.e. about 200 gallons).

The analyses for potential impact on groundwater quality due to the proposed landfill at Site F were performed using a lateral Darcy velocity of 0.05 m/a. This value was selected based on consideration of

- (i) hydrogeologic interpretation provided by Geocon; and
- (ii) a comparison of the calculated contaminant plumes for the Burlington landfill obtained for different values of v_a with the findings of the Burlington Landfill Plume Delineation Study (GLAL, 1986) as discussed in Section 6.

Assuming that the majority of the lateral flow is within a 25 m thick zone of fractured rock, the lateral Darcy velocity of 0.05 m/a corresponds to an average groundwater recharge of 1 mm/a over the 50 ha area of the landfill. This is the recharge of the "deep groundwater system" which might possibly carry contaminant beyond the site boundaries and excludes recharge of the upper weathered shale which will be collected by the leachate collection system. (It is noted in passing that similar results would be obtained if it was assumed that the recharge was 2 mm/a and the contaminant flowed through a 50 m thickness of fractured rock.)

THCL discuss dispersivity in their report (p. 98), and in their calculations they adopted dispersivities of 10% of the transport distance for weakly fractured shale. For the purposes of the analysis performed in the UW0 study of the proposed Halton landfill at Site F, the primary series of calculations was performed for a dispersivity of 30 m (cases 16, 17, 19, 20, 24 and 25) which is 10% of the travel distance of 300 m (i.e. to a point just south of Highway 403) and less than 4% of the travel distance of 800 m (i.e. to near the CNR tracks). To identify the effect of dispersivity, calculations were also performed for a dispersivity of 80 m which is less than 27% of the travel distance of 300 m and corresponds to 10% of the travel distance of 800 m.

In the analysis of the existing Burlington landfill, it was found that after 15 years the contaminant concentration calculated at a given point assuming a fracture spacing of 5 cm and 30 cm could differ quite substantially. The reason for this was that the time of interest (15 years) was relatively large compared to the time required for diffusion into the matrix to approach equilibrium for the 5 cm spacing (which takes less than one year) whereas this same time was substantially less than that required to approach equilibrium for a 30 cm spacing (which takes about 25 years). However, it should also be evident that the effect of different spacings should become less and less significant as the time of interest increases (i.e. particularly for time greater than 25 years). This is evident from Figs. 7 and 9 which show the concentration in the fractures at a distance of 300 m from the landfill for a range of times for "chloride" and "benzene" respectively. Results are given for two fracture spacings (viz. 30 cm and 5 cm). As can be seen, the difference between the results for the two fracture spacings is very small and is of no practical significance. This then

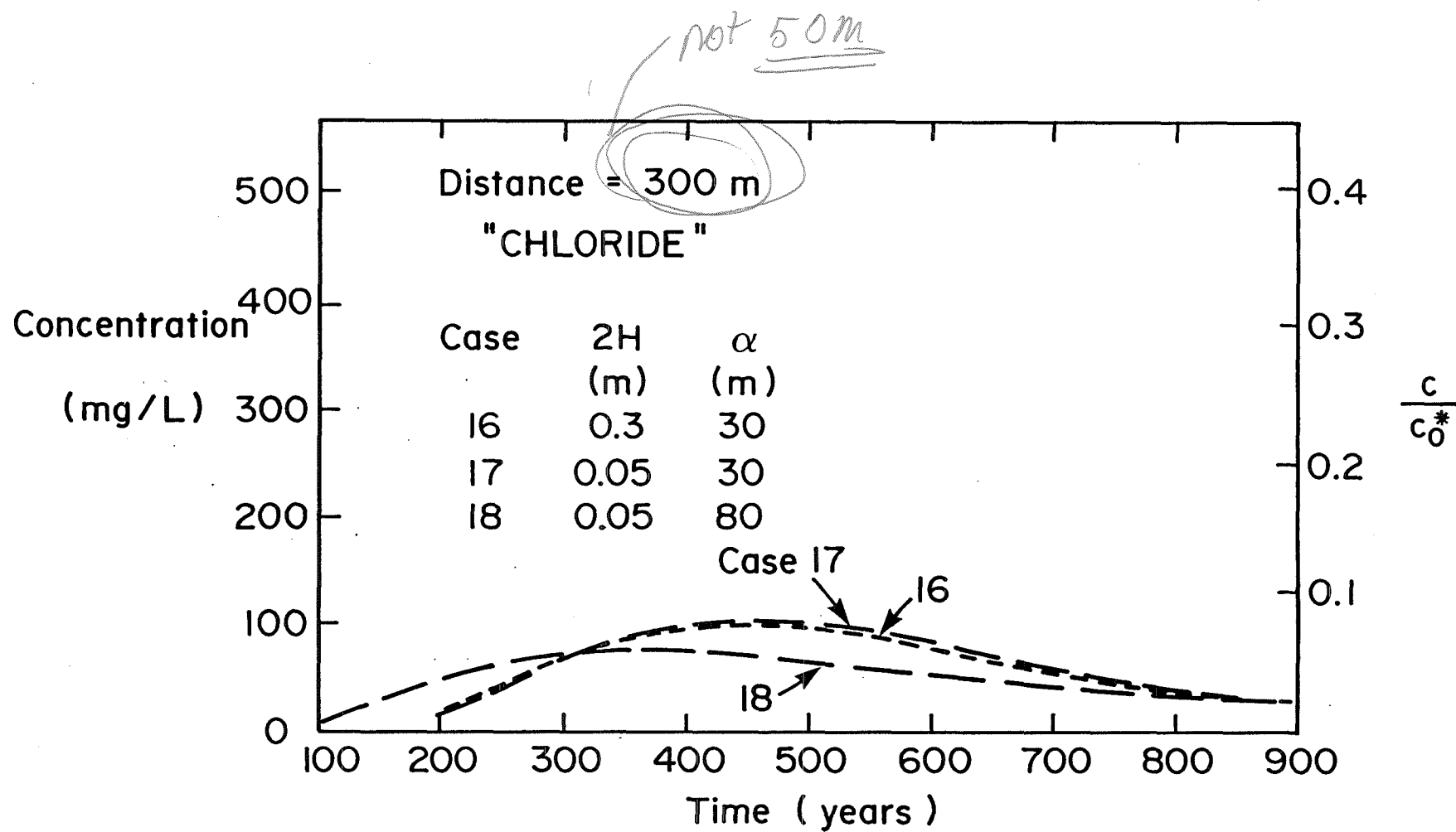


FIGURE 7 CALCULATED VARIATION IN "CHLORIDE" CONCENTRATION WITH TIME AT A POINT 300 m DOWNGRAIENT OF THE PROPOSED SITE F HALTON LANDFILL

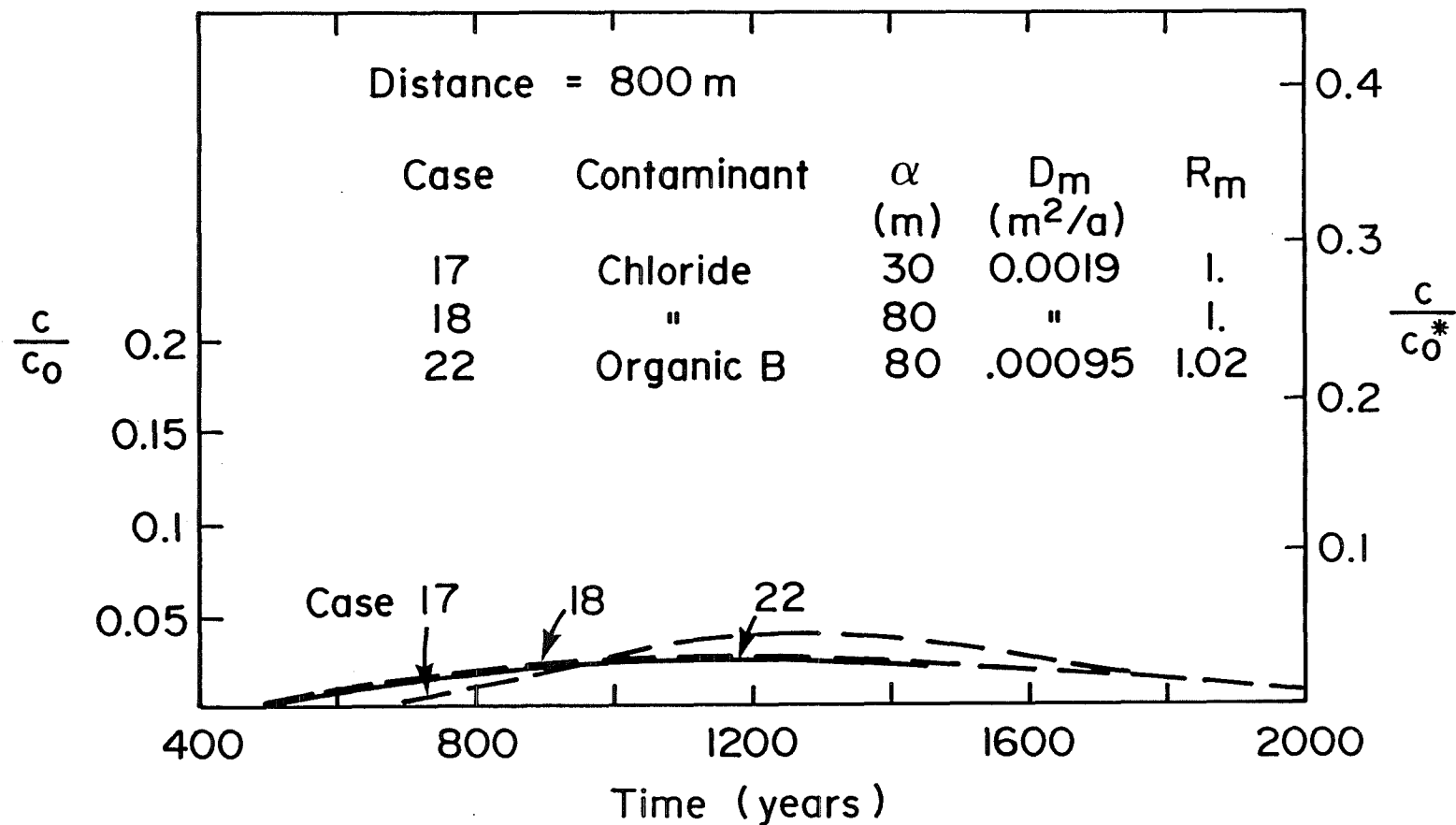


FIGURE 8 CALCULATED VARIATION IN CONTAMINANT CONCENTRATION WITH TIME AT A POINT 800 m DOWNGRADIENT OF THE PROPOSED HALTON LANDFILL (SITE F)

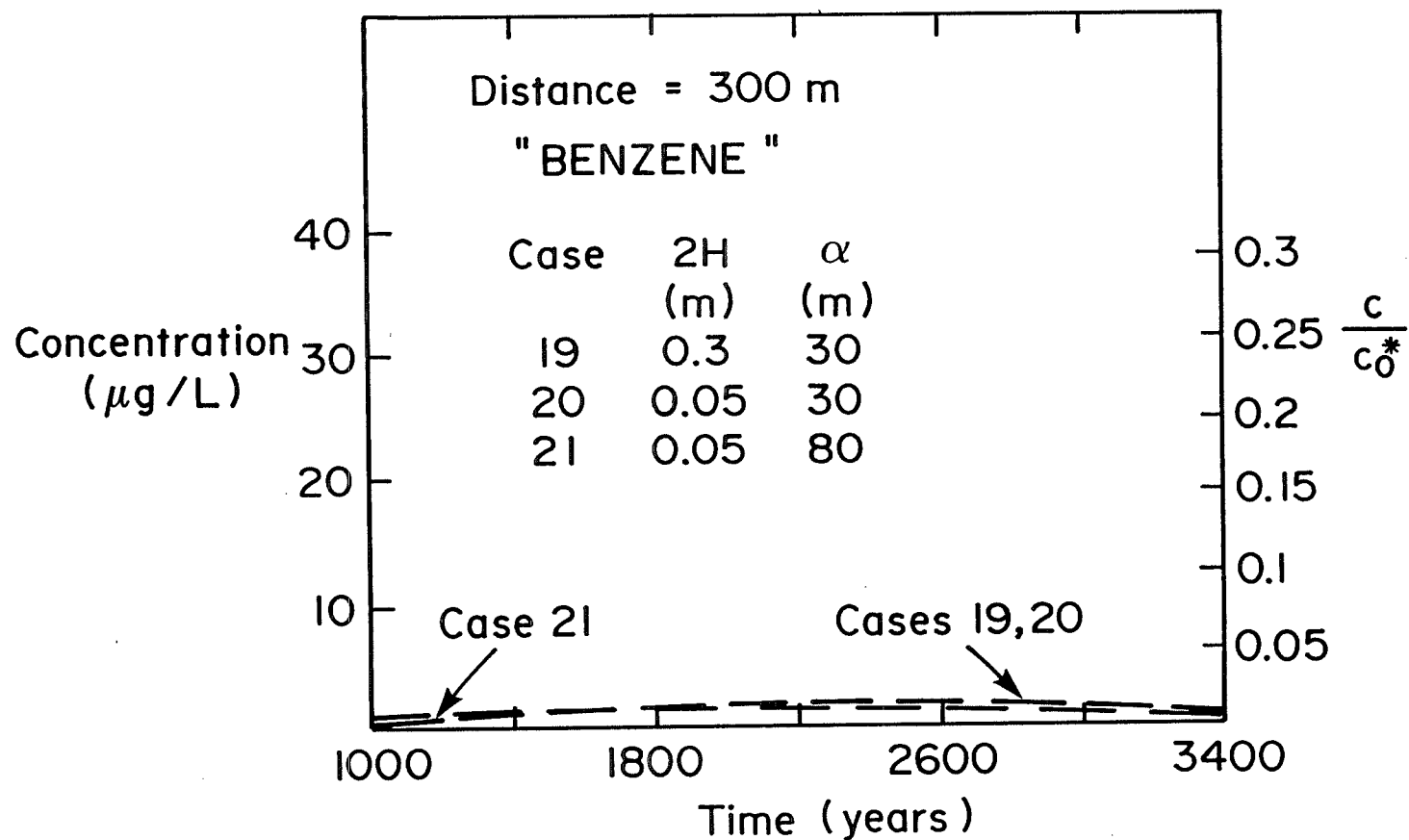


FIGURE 9 CALCULATED VARIATION IN "BENZENE" CONCENTRATION WITH TIME AT A POINT 300 m DOWNGRADIENT OF THE PROPOSED HALTON LANDFILL (SITE F)

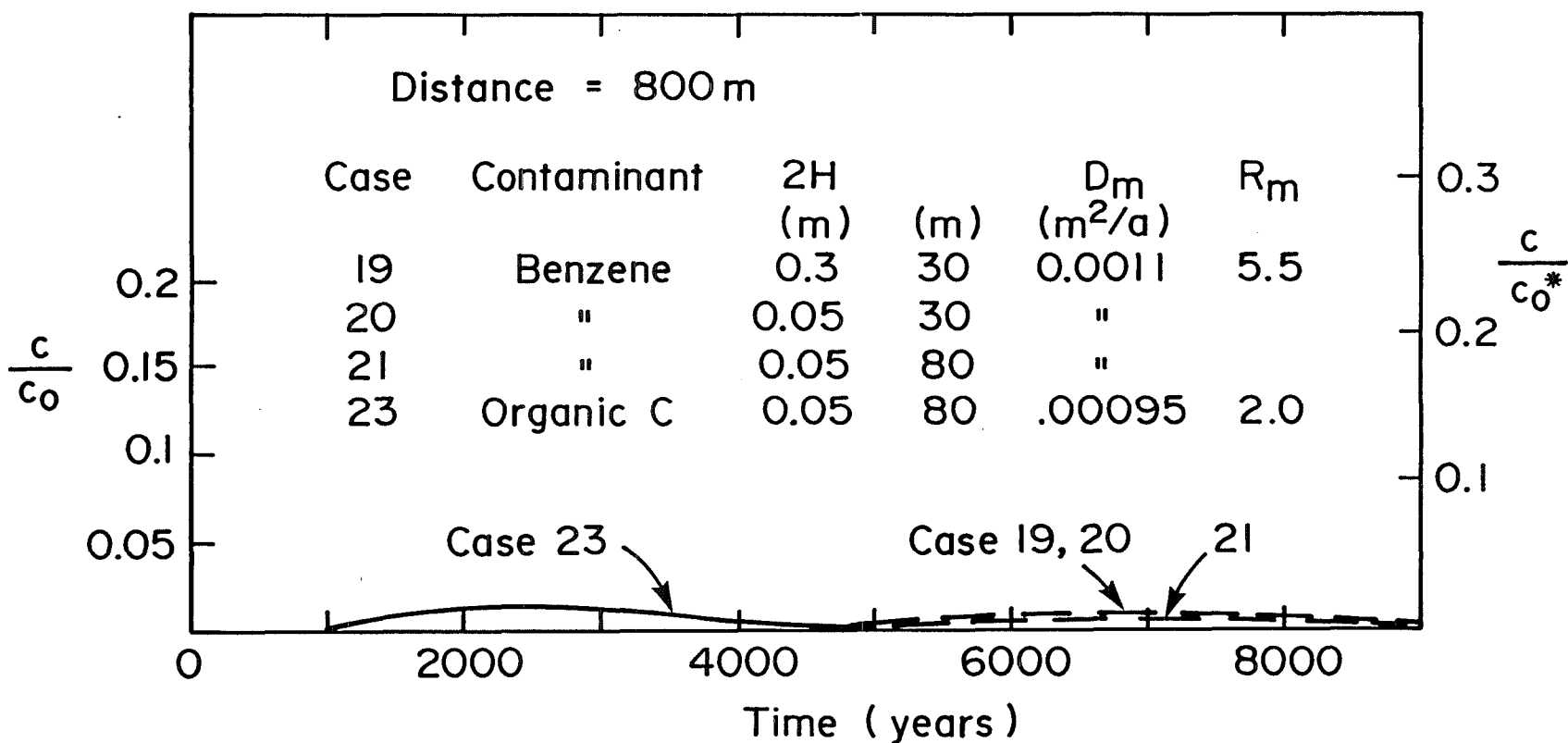


FIGURE 10 CALCULATED VARIATION IN ORGANIC CONTAMINANT CONCENTRATION WITH TIME AT A POINT 800 m DOWNGRADIENT OF THE PROPOSED HALTON LANDFILL (SITE F)

means that uncertainty regarding the exact spacing of the fractures ceases to be of practical significance when one is concerned with migration over long periods of time.

Examining Fig. 7, it can be seen that for a dispersivity of 30 m contaminant would reach concentrations exceeding 1% of the initial concentration in about 200 years and will continue to increase until the peak concentration, which is less than 7% of the initial concentration (i.e. about 100 mg/L), is reached after 450 years. If a higher dispersivity is assumed then contaminant reaches this point of interest faster but the peak concentration is reduced. For example in case 18, the peak concentration (at 300 m) of 5% of the initial concentration is reached in 350 years.

The level of attenuation increases with distance from the source. Figure 8 shows the calculated variation in "chloride" concentration at a point 800 m from the landfill. For a dispersivity equal to 4% of the travel distance ($\alpha = 30$ m, case 17) it would take about 1250 years to reach a peak "chloride" concentration of less than 60 mg/L or 4% of the initial concentration. With a dispersivity equal to 10% of the travel distance, it would still take about 1100 years to reach a very small peak value about 2.7% of c_0 .

As noted earlier, "chloride" is being considered as being representative of conservative species. In fact, the concentration of chloride itself which would actually be expected to occur will be much higher than the values calculated above. This is because the parent rock will be acting as the primary source of chloride (which will diffuse out from the rock matrix into the moving groundwater), rather than the landfill.

To illustrate that the level of attenuation obtained for "chloride" may be reasonable for other species, an analysis was performed for a highly mobile, volatile non-priority pollutant with a diffusion coefficient conservatively assumed

to be only half that of chloride (a value of $0.00095 \text{ m}^2/\text{a}$ was used) and a retardation factor R_m of only 1.02 (case 22 in Table 1). This contaminant (denoted as organic B) is not intended to be a specific contaminant but rather, represents a low (conservative) estimate of the diffusion coefficient and retardation factor for volatile organic pollutants. For example, the parameters for "organic B" (case 22) would be considered to be a conservative estimate of parameters for highly mobile contaminant such as methyl ethyl ketone (THCL quotes a retardation factor of 1.48 as being representative for this species--p. 105).

Figure 8 shows the calculated variation in the concentration of "organic B" (case 22) with time at a point 800 m from the landfill (i.e. near the CNR tracks) and it can be seen that to all practical purposes, the results are the same as those obtained for "chloride" using the same dispersivity. This indicates that when dealing with large travel times, the difference of a factor of two in the diffusion coefficient can have very little impact on the concentration of contaminant at points well away from the landfill. This is simply because the time scale for diffusion of contaminant into and out of the matrix is of the order of tens of years while we are concerned with time scales of hundreds to thousands of years for the peak contaminant concentration to be reached 800 m from the landfill.

Figures 9 and 10 show the variation in "benzene" concentration with time at points 300 m and 800 m from the landfill respectively. The general comment made above concerning the effects of fracture spacing and dispersivity apply equally well to "benzene" or any other contaminant species. The primary difference between the moderately sorbed contaminants such as benzene and conservative contaminant, such as chloride, is that sorption reduces the magnitude of the peak concentration and substantially increases the time to reach this peak concentration. For example in Fig. 9, the peak concentration of "benzene" at 300 m is

only about 1.5% of the initial concentration in the landfill and is not reached until 2200 years after construction of the landfill. A slightly smaller peak concentration is calculated for 800 m from the landfill and this peak would be reached in about 6500 years.

Analyses performed for a hypothetical contaminant (organic C) with a diffusion coefficient and retardation factor smaller than that for benzene (i.e. $D_m = 0.00095 \text{ m}^2/\text{a}$, $R_m = 2$) are also summarized in Fig. 10 (case 23). Comparing the results for cases 21 and 23, it can be seen that reducing the retardation factor from 5.5 to 2 reduced the time required to reach the peak concentration from 6000 years to about 2300 years but does not significantly increase the peak concentration which is a little over 1% of the initial concentration in the landfill.

8.0 WORST CASE SCENARIO - SITE F

Assuming total failure of the leachate collection tiles (but assuming that perimeter drains collect leachate in the shallow groundwater), analyses were performed for an average flow per unit area (Darcy velocity) of 0.125 m/a. This is based on the very conservative assumption that leachate mounding would occur to the top of the landfill and hence the gradient between the toe of the landfill and a point 300 m south would increase by a factor of 2.5. It is assumed that the flow could be accommodated by the flow system further south. The calculated variation in concentration with time at 300 m and 800 m from the landfill for chloride (Case 24) and benzene (Case 25) are shown in Figures 11 and 12 respectively.

From Figure 11 it is seen that even for this "worst case" the peak concentration of a conservative species at a point 300 m south of the landfill only

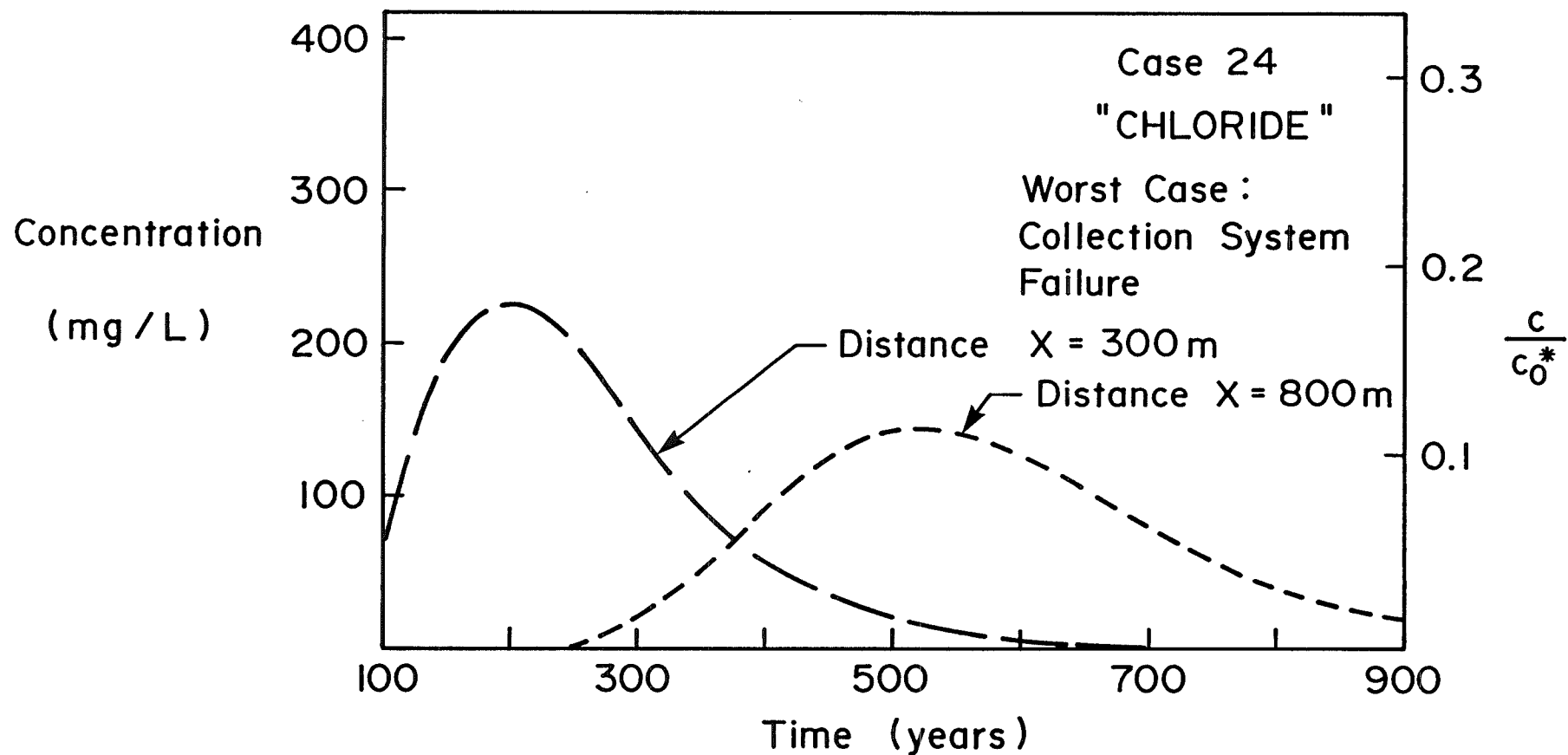


FIGURE 11 CALCULATED VARIATION IN "CHLORIDE" CONCENTRATION WITH TIME DOWNGRADIENT OF THE PROPOSED HALTON LANDFILL (SITE F) - "WORST" CASE SCENARIO

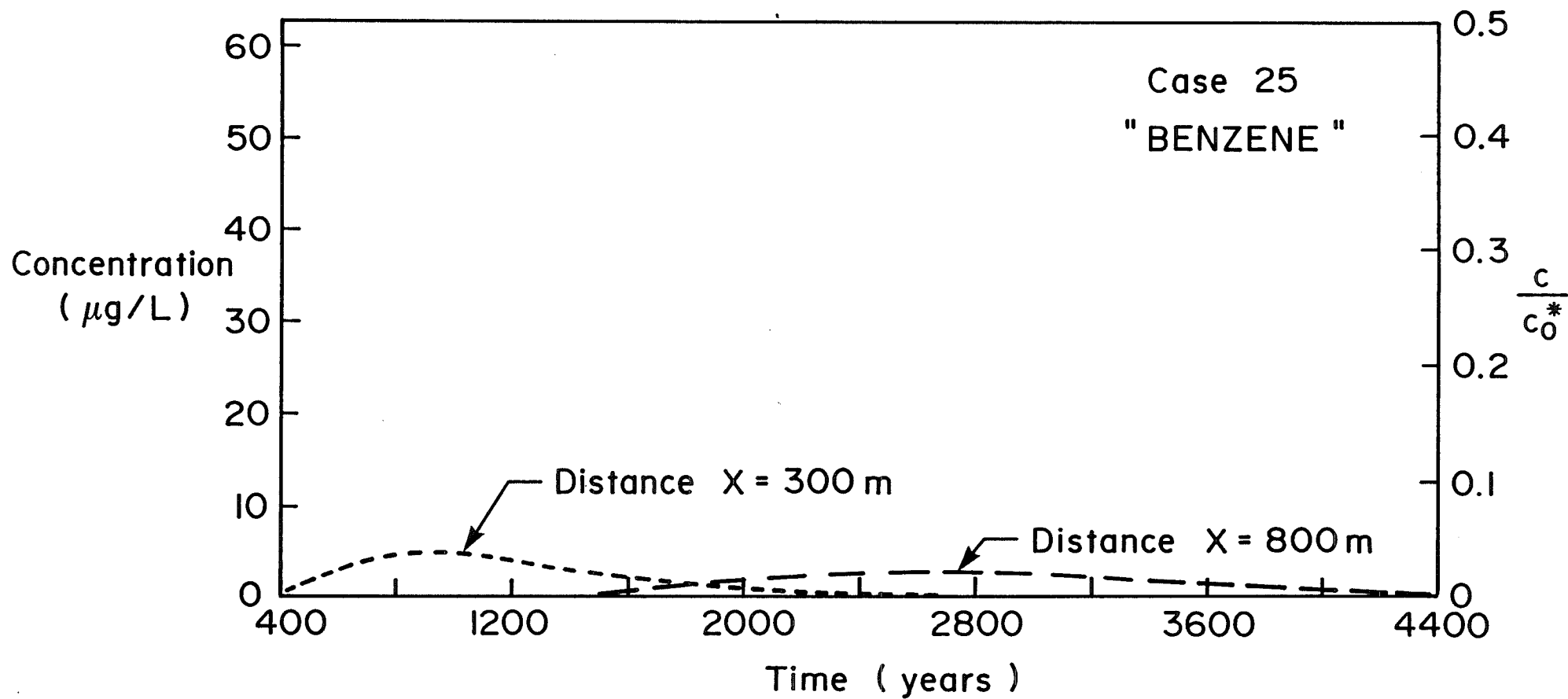


FIGURE 12 CALCULATED VARIATION IN "BENZENE" CONCENTRATION WITH TIME DOWNGRADIENT OF THE PROPOSED HALTON LANDFILL (SITE F) - "WORST" CASE SCENARIO

reaches 15% of the initial leachate concentration, c_0 , after 200 years. By the time the contaminant has moved 800 m, the peak concentration has reduced to less than 10% of c_0 is not reached until about 525 years after the failure occurs.

As might be expected, for a retarded species such as benzene, even lower concentrations are predicted (see Fig. 12) and at 300 m the peak of about 3% of c_0 is not expected to occur until 900 years after failure of the collection system. At 800 m it would take 2700 years to reach a peak concentration of less than 2% of c_0 .

9.0 CONCLUSION

Due to uncertainty as to the actual values of many key parameters, no analysis performed for a specific set of parameters could be regarded as providing a prediction of the precise concentrations and arrival times to be expected in reality. In circumstances such as these, it is necessary to assess the potential impact of the proposed landfill by considering the results of analyses performed for combinations of parameters which, on the basis of engineering judgement, might reasonably be expected to occur.

A number of laboratory tests were performed to determine the diffusion coefficient for chloride in the Queenston shale from Site F. To be conservative, the diffusion coefficient determined from the laboratory tests was reduced by a factor of 2.5 giving a conservative value of $0.0019 \text{ m}^2/\text{a}$ ($6 \times 10^{-7} \text{ cm/s}$). This value is generally consistent with the value of $5 \times 10^{-7} \text{ cm/s}$ assumed by THCL in their studies for Site F.

Analyses were performed to estimate the contaminant escape from the existing Burlington landfill after 15 years. A range of fracture spacings, fracture opening sizes, fracture porosities and average linearized groundwater velocities

were examined both for a conservative species such as chloride and an organic priority pollutant such as benzene. The results of these studies indicate that an extensive and readily detectable contaminant plume would have developed downgradient of the existing Burlington landfill if in fact the average lateral flow per unit area were about 0.7 m/a (which corresponds to an average recharge of 14 mm/a calculated by THCL from their initial finite element analysis). Irrespective of whether one was considering conservative species or organics such as benzene, the presence of the plume should have been evident to any investigator. The very fact that there can be any debate about whether a plume has or has not developed south of the landfill is a clear indication that the average lateral flow in the fractured rock is considerably less than 0.7 m/a.

Analyses of contaminant migration from the Burlington landfill performed for an average lateral flow per unit area of 0.125 m/a (corresponding to the worst case mounding envisaged for Site F) indicated that for these assumed conditions, conservative contaminants would be readily detected 25 m from the landfill (i.e. at GL 12 and 24) and might be detected at very low concentrations as far south as GL 16. Organic contaminants such as benzene would be detectable (at the 1 µg/L level) at 25 m from the landfill if the average fracture spacing was about 30 cm but would probably not be detected if the fracture spacing was about 5 cm. This scenario could be approached if there was significant leachate mounding of contaminants in the portion of the existing Burlington landfill which does not have a leachate underdrain system.

Analyses of contaminant migration from the existing Burlington landfill, performed assuming an average lateral flow per unit area of 0.05 m/a, indicated that conservative contaminants would be expected to have moved 25 m from the landfill but that organics such as benzene probably would not have extended this

far in 15 years since the landfilling commenced. This appears to be consistent with the findings of the Burlington Landfill Plume Delineation Study (GLAL, 1986). Furthermore, the velocity of 0.05 m/a is consistent with hydrogeologic interpretation provided by Geocon.

Migration from the proposed Halton landfill at Site F was modelled for the average lateral flow per unit area of 0.05 m/a. These analyses indicate that 300 m downgradient from the landfill (i.e. just south of Highway 403), the peak concentration of conservative species and a moderately sorbed species is likely to be less than 7% and 2%, respectively, of the maximum concentration in the landfill. The time required to reach these small peak concentrations ranges from hundreds to thousands of years. At a point 800 m downgradient of the landfill (near the CNR tracks), the peak concentration is even smaller and the time for the peak to occur is of the order of thousands of years.

Based on the available data, there is uncertainty regarding factors such as the spacing of fractures, the fracture opening size and the dispersivity along the fractures. However, an examination of the effect of a reasonable variation in these parameters indicates that within the range considered, the uncertainty has no significant effect on the peak concentrations at the two points of interest (300 m and 800 m). The greatest uncertainty in this respect is the dispersivity and even using a very low dispersivity of less than 4% of the travel distance, the peak concentrations at the 800 m point did not exceed 5% of the initial concentration. Increasing the dispersivity decreases the peak concentration and the time required to reach the peak. Adopting a fairly high dispersivity equal to 27% of the travel distance gave times of 350 years to reach the peak concentration at a point 300 m from the landfill.

Even considering a worst case scenario involving leachate mounding to the

top of the waste and a consequent increase in average lateral flow per unit area to 0.125 m/a, the peak concentration of conservative species was estimated to be less than 15% of the initial concentration, c_0 , at 300 m from the landfill, reducing to less than 10% of the initial concentration at 800 m from the landfill. The times required to reach these concentrations are 200 and 525 years respectively.

It should also be noted that the contaminant studies reported herein have not taken account of the beneficial effects of dilution of contaminant due to recharge of the rock from northwest of the proposed Site F. Nor have the potentially beneficial effects of attenuation mechanisms such as biodegradation and microbial transformation of organic contaminants discussed by THCL (Vol. 1, pp. 85-86) been considered. Thus the concentrations of contaminant calculated in this study are considered to be conservative.

The findings of the Burlington Landfill Plume Delineation Study (GLAL, 1986) southeast of the Burlington landfill are consistent with the results of analyses performed in the UWO study for an anticipated average lateral flow per unit area of 0.05 m/a (which corresponds to a fracture velocity of 50 m/a for a fracture porosity of 0.001). Based on the results of analyses performed for the proposed Halton landfill at Site F using an average lateral flow per unit area of 0.05 m/a, it is concluded that there would be minimal effect on the groundwater quality southeast of Highway 403 due to contaminant migration in the deep groundwater system from Site F.

Even if one adopts the worst case scenario for the extreme situation of mounding to the top of the waste, the potential impact on groundwater quality southeast of Highway 403 is relatively small.

As noted above, the proposed landfill will have minimal impact on deep groundwater quality. Thus, in the event that there was a discharge of this groundwater into the shallow groundwater system southeast of Highway 403, the impact of the landfill on this shallow groundwater would also be minimal. Furthermore, upward movement of deep groundwater through the till would result in additional attenuation and improvement in the deep groundwater quality. Indeed, in the event that the deep groundwater did discharge in this manner, one would expect the impact of the naturally poor quality of this groundwater to be far more significant than any impact due to contaminant in this groundwater which originated from the proposed Halton landfill.

Comparing the results of analyses for both Sites D and F, it is concluded that the potential for significant groundwater contamination southeast of Highway 403 due to contaminant migration from the proposed Site F is minimal compared to the likelihood of contamination of aquifers and residential water supplies adjacent to the proposed Site D.

The analyses performed in this study and the conclusions reached are based on the assumption that the Proposed Halton Landfill at either Site F or D is for the disposal of municipal waste. No consideration has been given to the implications of disposing of quantities of hazardous/toxic waste over and above the levels normally encountered in municipal leachate.

10. REFERENCES

- Barker, J.A. and Foster, S.S.D. (1981). A diffusion exchange model for solute movement in fissured porous rock. Q. J. Eng. Geol., London, Vol. 14, pp. 17-24.
- Chiou, C.T. and Schmedding, D.W. (1982). Partitioning of organic compounds in octanol-water systems. Journal of Environmental Science and Technology, Vol. 16, p. 4.
- Day, M.J. (1977). Analysis of movement and hydrochemistry of groundwater in the fractured clay and till deposits of the Winnipeg area, Manitoba. M.Sc. Thesis, University of Waterloo, p. 210.
- Freeze, R.A. and Cherry, J.A. (1979). Groundwater. Englewood Cliffs, New Jersey: Prentice-Hall Inc.
- Foster, S.S.D. (1975). The chalk groundwater tritium anomaly--A possible explanation. Journal of Hydrology, Vol. 25, pp. 159-165.
- Gillham, R.W. and Cherry, J.A. (1982). Contaminant migration in saturated unconsolidated geologic deposits. Geophysical Society of America, Special Paper 189, pp. 31-62.
- GLAL (Garnter Lee and Associates Limited) (1984). Hydrogeological Study: Burlington Landfill Site Continued Use For M.M. Dillon Ltd.), p. 0-31.
- GLAL (Gartner Lee and Associates Limited) (1986). Burlington Regional Landfill Plume Delineation Report for the Regional Municipality of Halton. See also: Halton Regional Landfill - Burlington. 1986 Monitoring Report.
- Grisak, G.E. and Pickens, J.F. (1980). Solute transport through fractured media, 1, The effect of matrix diffusion. Water Resources Research, 16(4), pp. 719-730.

- Grisak, G.E., Pickens, J.F. and Cherry, J.A. (1980). Solute transport through fractured media, 2, Column study of fractured till. *Water Resources Research*, Vol. 16, pp. 731-739.
- Hewetson, J.P. An investigation of the groundwater zone in fractured shale at a landfill. M.E.Sc. Thesis, University of Waterloo, p. 27.
- Hoek, E. and Bray, J.W. (1981). *Rock Slope Engineering*, 3rd Ed. The Institute of Mining and Metallurgy, London, p. 133.
- Reid, R.C., Prausnitz, J.M. and Sherwood, T.K. (1977). *The Properties of Gases and Liquids*. New York: McGraw-Hill.
- Schwarzenbach, R.P. and Westall, J. (1981). Transport of nonpolar organic compounds from surface water to groundwater. Laboratory sorption studies. *Journal of Environmental Science and Technology*, Vol. 15, p. 1360.
- Sudicky, E.A. and Frind, E.O. (1982). Contaminant transport in fractured porous media: analytical solutions for a system of parallel fractures. *Water Resources Research*, 18(6), pp. 1634-1642.
- Talbot, A. (1979). The accurate numerical integration of Laplace Transforms. *J. Inst. Maths. Applics.*, 23, pp. 97-120.
- Tang, D.H., Frind, E.O. and Sudicky, E.A. (1981). Contaminant transport in fractured porous media: analytical solution for a single fracture. *Water Resources Research*, 17(3), pp. 555-564.

APPENDIX A: Theory (Rowe & Booker, to be published)

INTRODUCTION

The theoretical model used in the evaluation of the potential for contaminant migration in the fractured Queenston shale at Site F represents an extension of earlier models proposed by Tang et al. (1981), Barker and Foster (1981), and Sudicky and Frind (1982) insofar as it is assumed that the contaminant transport can be regarded as one dimensional along the fractures and one dimensional into the matrix perpendicular to the fractures. However, in this present formulation, it is not assumed that concentration of contaminant in the source remains constant, but rather, the more realistic assumption of finite mass of contaminant is adopted. An analytic solution is then obtained in transform space and it simply remains to numerically invert the Laplace Transform. Using a modern algorithm for inverting the Laplace Transform, it turns out that this approach is both less restrictive and numerically far more efficient than the Sudicky and Frind solution where they analytically invert the Laplace Transform and then numerically evaluate two integrals (assuming constant source concentration).

ASSUMPTIONS

Considering the situation shown in Fig. 1, it is assumed that the contaminant is transported vertically downward into the rock beneath the landfill and can then move horizontally through a series of parallel fractures of width $2b$ which have a spacing $2H$. In parallel with earlier formulations (eg. Sudicky and Frind, 1982) it is assumed that:

1. The width of each fracture is much smaller than its length;

2. Transverse diffusion and dispersion within each fracture assures complete mixing across its width at all times;
3. The permeability of the intervening porous matrix is low and transport within the matrix is mainly by molecular diffusion; and
4. Transport along each fracture is much faster than transport within the matrix.

THEORY

It will be assumed that the fissures are parallel to the xy plane and that the fluid in the fissure moves with a velocity v_f in the x direction.

Neglecting possible sorption on the face of the cracks, the equation for advective-dispersive transport of the contaminant along a typical fissure is:

$$n_f D_f \frac{\partial^2 c_f}{\partial x^2} - n_f v_f \frac{\partial c_f}{\partial x} = n_f \frac{\partial c}{\partial t} - \frac{F}{b} \quad (A1)$$

where c_f is the concentration of the contaminant within the fissure

n_f is the porosity of the fissure material. If the fracture is "open"

(i.e. free of material impeding flow along the fracture) then $n_f=1$

D_f is coefficient of hydrodynamic dispersion (incorporating the effects

of molecular diffusion and mechanical dispersion) in the fissure

v_f is the average linearized groundwater velocity in the fracture

F is the flux leaving the fissure and moving into the matrix.

It will be assumed that there is no advection in the matrix and that, to sufficient approximation, all diffusion occurs in the vertical (z) direction. It then follows that

$$n_m D_m \frac{\partial^2 c_m}{\partial z^2} = n_m \frac{\partial c_m}{\partial t} + \rho_m K_d \frac{\partial c_m}{\partial t} \quad (A2)$$

where c_m is the concentration of contaminant in the matrix

n_m is the porosity of the matrix

D_m is the coefficient of molecular diffusion in the matrix

ρ_m is the dry density of the rock matrix

K_d is a distribution or partitioning coefficient for removal of contaminant from solution due to linear sorption.

Initially, the concentration in the matrix is zero and so introducing the Laplace Transform

$$\bar{c}_m = \int_0^\infty c_m(t) e^{-st} dt \quad (A3)$$

it is found that

$$\frac{\partial^2 \bar{c}_m}{\partial z^2} = \mu^2 \bar{c}_m \quad (A4a)$$

where

$$\mu^2 = (1 + \rho_m K_m / n_m) s / D_m$$

$$\text{i.e., } \mu^2 = R_m s / D_m \quad (A4b)$$

$$\text{where } R_m = 1 + \rho_m K_m / n_m \quad (A4c)$$

It is convenient to locate the origin of coordinates midway through the matrix then recognizing that the matrix and fissure concentrations must be identical at their interface it follows that

$$\bar{c}_m = \bar{c}_f \frac{\cosh \mu z}{\cosh \mu H} \quad (A5)$$

The flux χ entering the fissure may now be calculated and

$$\bar{F} = -n_m D_m \frac{\partial \bar{c}_m}{\partial z} \text{ when } z = H \quad (A6a)$$

so that

$$\bar{F} = -\chi \bar{c}_m \quad (A6b)$$

where $\chi = n_m D_m \mu \tanh \mu H$.

Initially, the concentration in the fissure is zero and thus, taking the Laplace Transform of equation (1) and combining it with equation (6b), it is found that

$$D_f \frac{\partial^2 \bar{c}_f}{\partial x^2} - v_f \frac{\partial \bar{c}_f}{\partial x} = (s + \frac{\chi}{n_f b}) \bar{c}_f \quad (A7)$$

It will be assumed that the length of each fissure is far greater than its thickness and thus that solution of (7) must vanish as $x \rightarrow \infty$ so that

$$\bar{c}_f = \bar{c}_{LF} e^{-\lambda x} \quad (A8)$$

where c_{LF} denotes the concentration in the landfill and

$$\lambda = \frac{1}{2D_f} \{-v_f + (v_f + 4D_f(s + \frac{\chi}{n_f b}))^{1/2}\} \quad (A9)$$

It remains to compute the concentration in the landfill. To do this, we introduce the equivalent height of leachate per fissure (H_{LF}) as defined in the report. It then follows from conservation of mass that

$$H_{LF}[c_{LF}(0)-c_{LF}(t)] = - \int_0^t n_f(D_f \frac{\partial c_f}{\partial x} - v_f c_f) dt \quad (A10)$$

where the integral on the right-hand side of this equation is evaluated at $x=0$, so that upon Laplace transformation we find

$$\bar{c}_{LF} = \frac{c_{LF}(0)}{s+n_f(D_f \lambda + v_f)/H_{LF}} \quad (A11)$$

The complete solution in Laplace Transform space is given by equations (A8) and (A11). The Laplace Transform can be inverted using Talbot's algorithm (Talbot, 1979).

APPENDIX B: Results of Additional Calculations Relating to the
Burlington Landfill

Figures 4 and 5 of the main report indicate the extent of the contaminant plume that is calculated after 15 year migration assuming a lateral flow per unit area (Darcy velocity) of 0.05, 0.125 and 0.7 m/a. The appendix contains results calculated after 10 years and 25 years based on the same assumptions as adopted in Figs. 4 and 5. Specifically, Figures 13 and 15 show the calculated extent of a benzene plume after 10 and 25 years, respectively, while Figures 14 and 16 show the extent of movement of a conservative species such as chloride after 10 and 25 years, respectively.

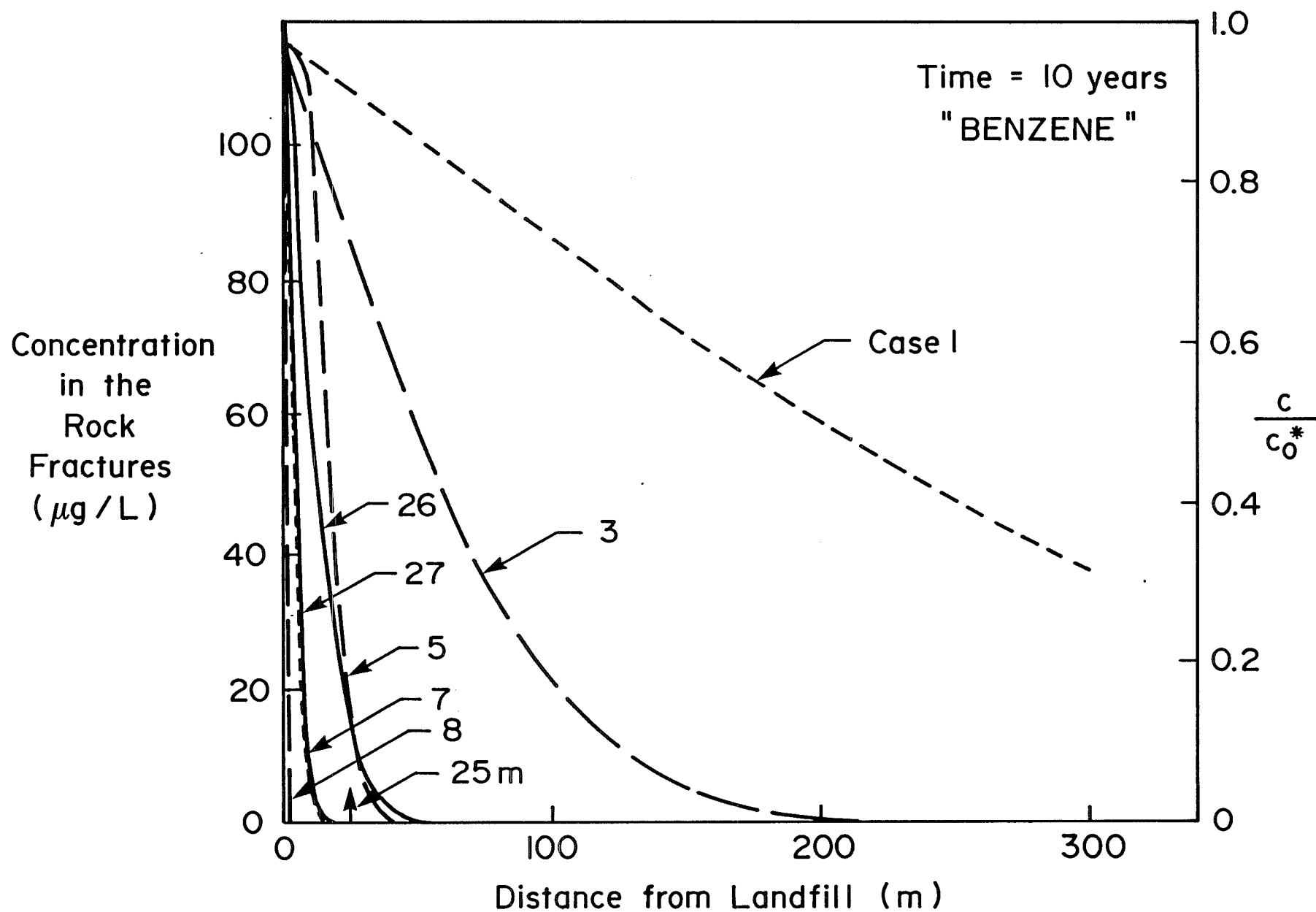


FIGURE 13 CALCULATED MIGRATION OF "BENZENE" FROM THE BURLINGTON LANDFILL AFTER 10 YEARS

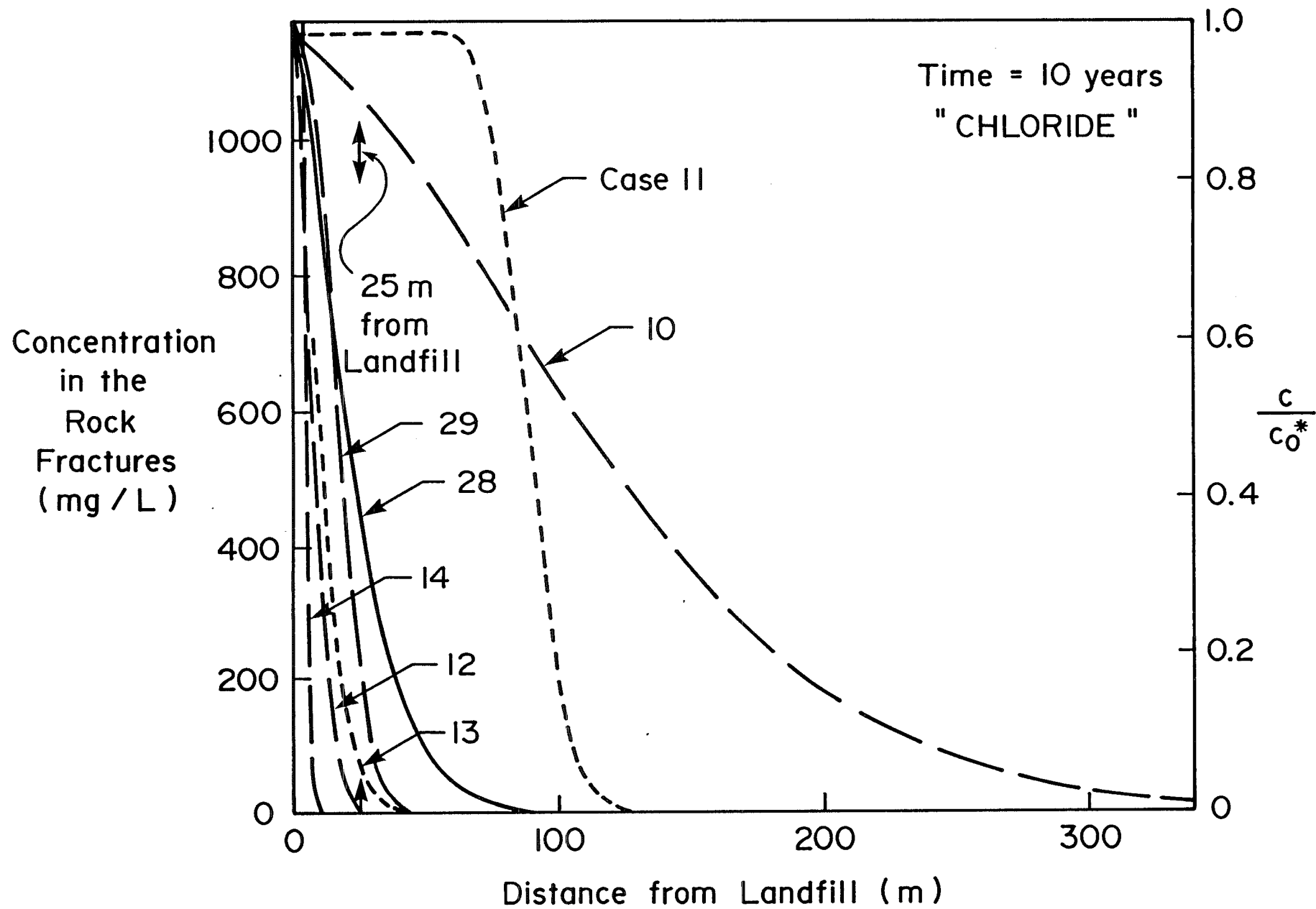


FIGURE 14 CALCULATED MIGRATION OF "CHLORIDE" FROM THE BURLINGTON LANDFILL AFTER 10 YEARS

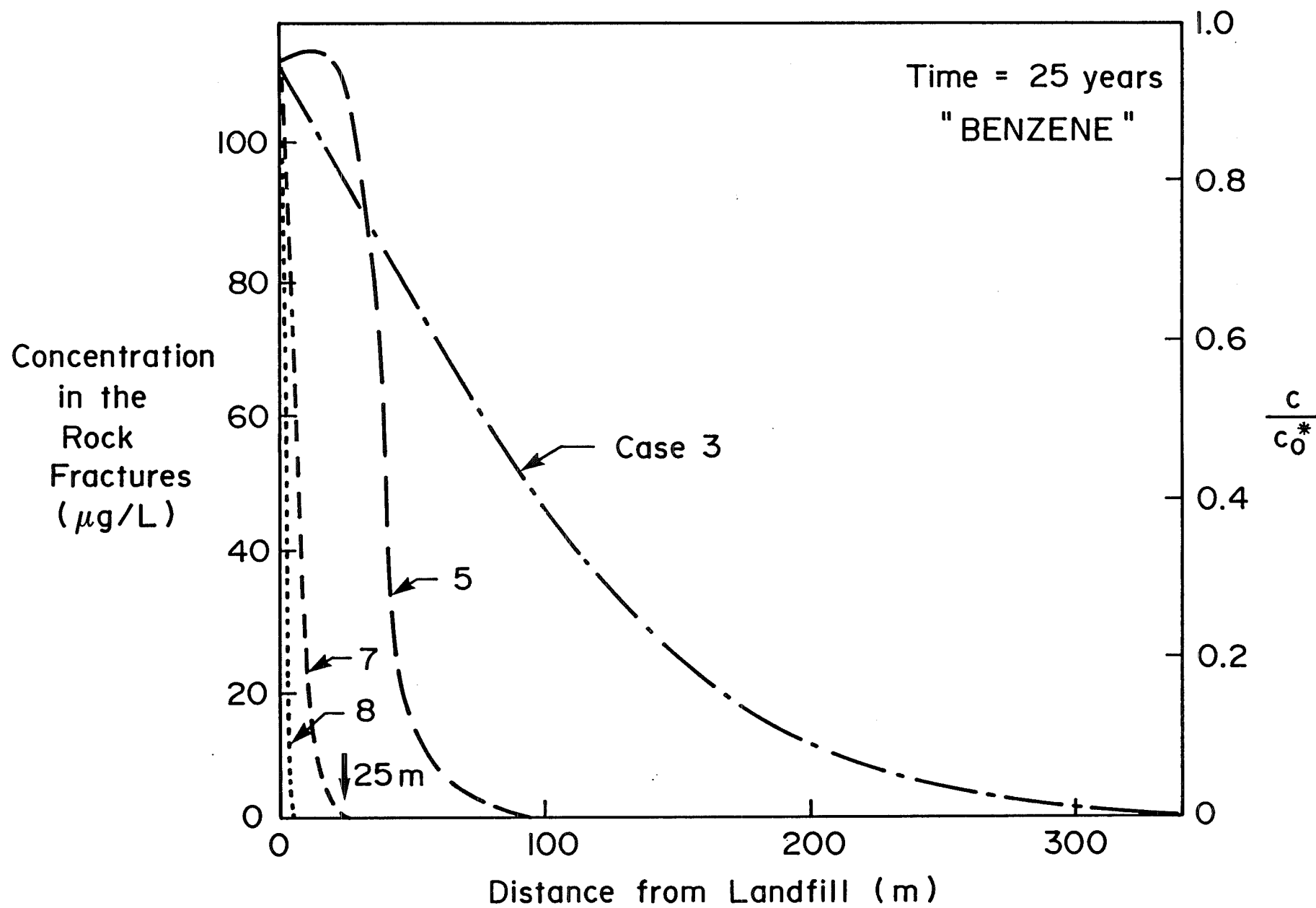


FIGURE 15 CALCULATED MIGRATION OF "BENZENE" FROM THE BURLINGTON LANDFILL AFTER 25 YEARS

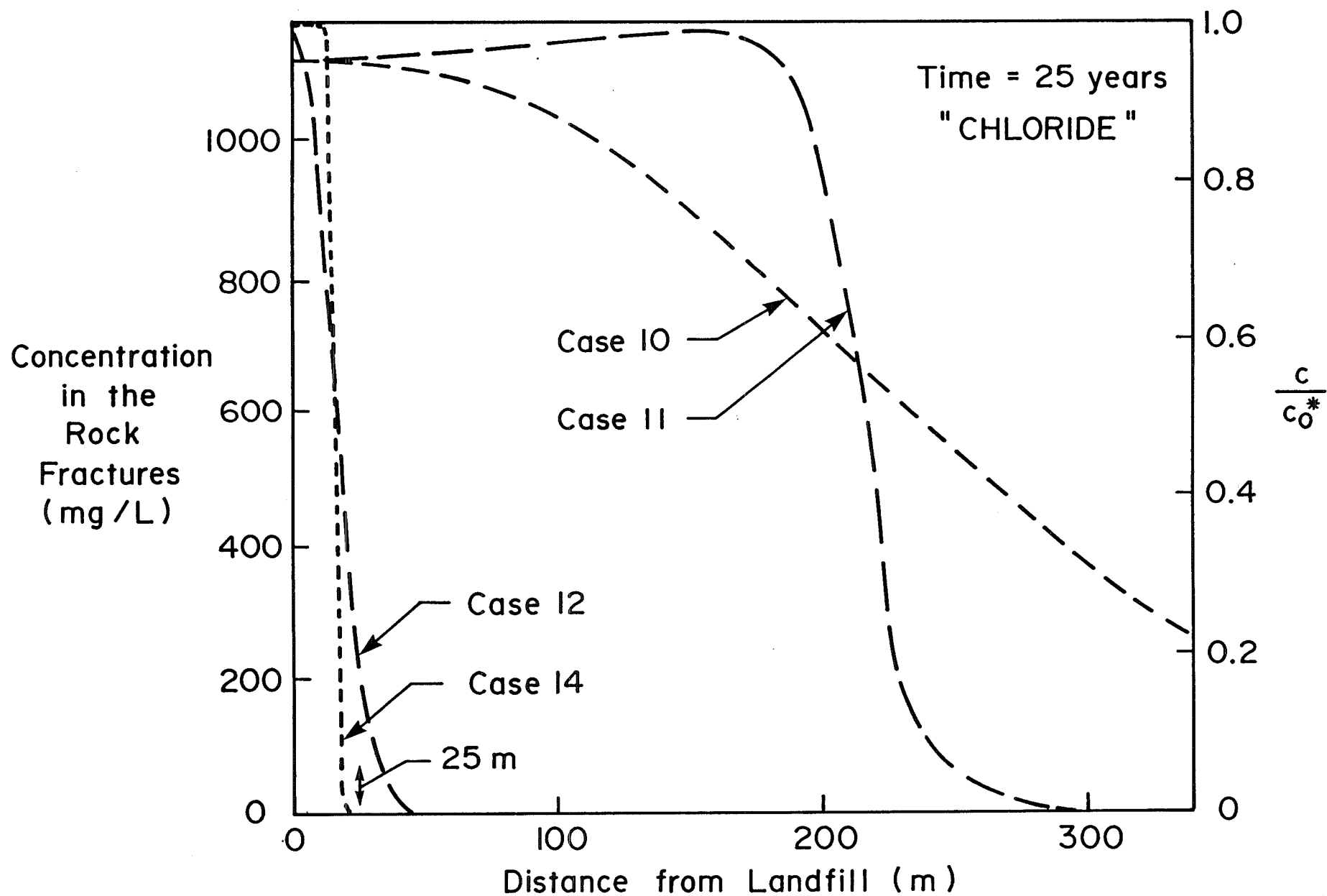


FIGURE 16 CALCULATED MIGRATION OF "CHLORIDE" FROM THE BURLINGTON LANDFILL AFTER 25 YEARS

APPENDIX C: Analysis of Leaching Profiles for Chloride at Site F

The transport model adopted in this report has assumed that the potential path for contaminant transport within the deep groundwater involves infiltration of about 1 mm/a down through the vertical fractures in the rock and then lateral flow offsite. (This excludes infiltration into the shallow groundwater system which will be collected by the leachate collection system.)

THCL originally envisaged (see Vol. 1, p. 62) recharge down vertically to a depth in excess of 25 m over most of Site F followed by lateral flow offsite. This scenario involved infiltration into the deep groundwater system of approximately 14 mm/a (Vol. 1, p. 63). THCL's so-called "Better Fit Model" dated September 1987, involves downward flow for the deep groundwater system over the southern portion of the site and what appears to be a substantially decreased average downward flow compared with their original calculations.

Assuming that prior to the establishment of the current flow system there was a relatively uniform distribution of chloride within the Queenston Shale deposit, then flow downward through the shale after development of the flow system would be expected to cause leaching of the chloride from the pores of the rock matrix. This is in fact the reverse of the matrix diffusion process discussed in the report. If water entering the fracture system has negligible salt concentration initially, then strong diffusion gradients from the rock matrix into the fractures would be established near the surface of the rock. As the water moves through the fractures there would be an increase in salt content and hence a lower diffusion gradient would be developed at depth. This in turn would be expected to give rise to a concentration profile both within the pore water and the rock involving low concentrations at the surface and increasing concentration

with depth. The magnitude of the concentrations would be related to the vertical flow within the fractured system and the length of time over which this flow has been occurring.

To provide a quantitative assessment of the mechanism, analyses were performed to model this process. The analyses were performed using a diffusion coefficient for chloride diffusing out of the shale determined, from three tests conducted at UW0, to be $0.0047 \text{ m}^2/\text{a}$ and a matrix porosity of 0.1. The initial concentration of chloride in the rock matrix was assumed to be 40 000 mg/L (the calculated concentrations are directly proportional to this quantity) and a dispersivity of 2.5 m was adopted.

The concentration profiles calculated assuming a fracture spacing of 1 m and a fracture opening size of 2 mm are shown in Fig. 17 for a range of times and two assumed infiltration rates, namely 1 mm/a and 14 mm/a. The concentration profile is highly dependent on the infiltration.

At an infiltration of 14 mm/a (corresponding to THCL's original flow system), concentrations of less than 7000 mg/L would be expected to a depth of about 30 m after only 250 years of flow and after 500 years the concentrations to a depth of 30 m would be less than 780 mg/L.

An infiltration rate of 1 mm/a as assumed in this study would give concentrations of less than 7450 mg/L after 3500 years and it would take 6000 years before the concentration would be below 1300 mg/L (at all points) to a depth of 30 m.

The effect of fracture spacing and fracture porosity was examined for fracture spacing ranging between 5 cm and 100 cm and fracture porosities ranging between 0.001 and 0.01. A number of typical results are shown in Fig. 18 for an infiltration of 1 mm/a (after 2000 years). Inspection of Fig. 18 shows that the effect of fracture spacing and porosity is of no practical significance. One can

conclude from this that within the range of parameters considered, the variation in chloride concentration with depth is primarily dependent on the infiltration (i.e. the Darcy velocity) and the time over which this infiltration has been occurring.

It is recognized that in a practical situation the infiltration will not be identical at all points over the site and that in some areas the infiltration may be larger, and in some areas smaller, than the average infiltration. This will, of course, result in a scatter of concentration profiles determined at different locations however one would expect the trend established by combining representative data from over a given site to follow the calculated trend for the average infiltration. Local variations in concentration profile may represent local variations in infiltration. Thus if, for example, one considered that the average infiltrations of 1 mm/a had been established for a period of about 3000 years, then if locally the infiltrations were 2 mm/a you would expect the concentration profile at that location to approximately correspond to that shown on Fig. 17 for an infiltration of 1 mm/a and a time of 6000 years. Similarly, if at another location the infiltration was half the average (i.e. 0.5 mm/a) then one would expect the concentration profile at that location to be similar to that given in Fig. 17 for $v_a = 1$ mm/a and a time of 1500 years.

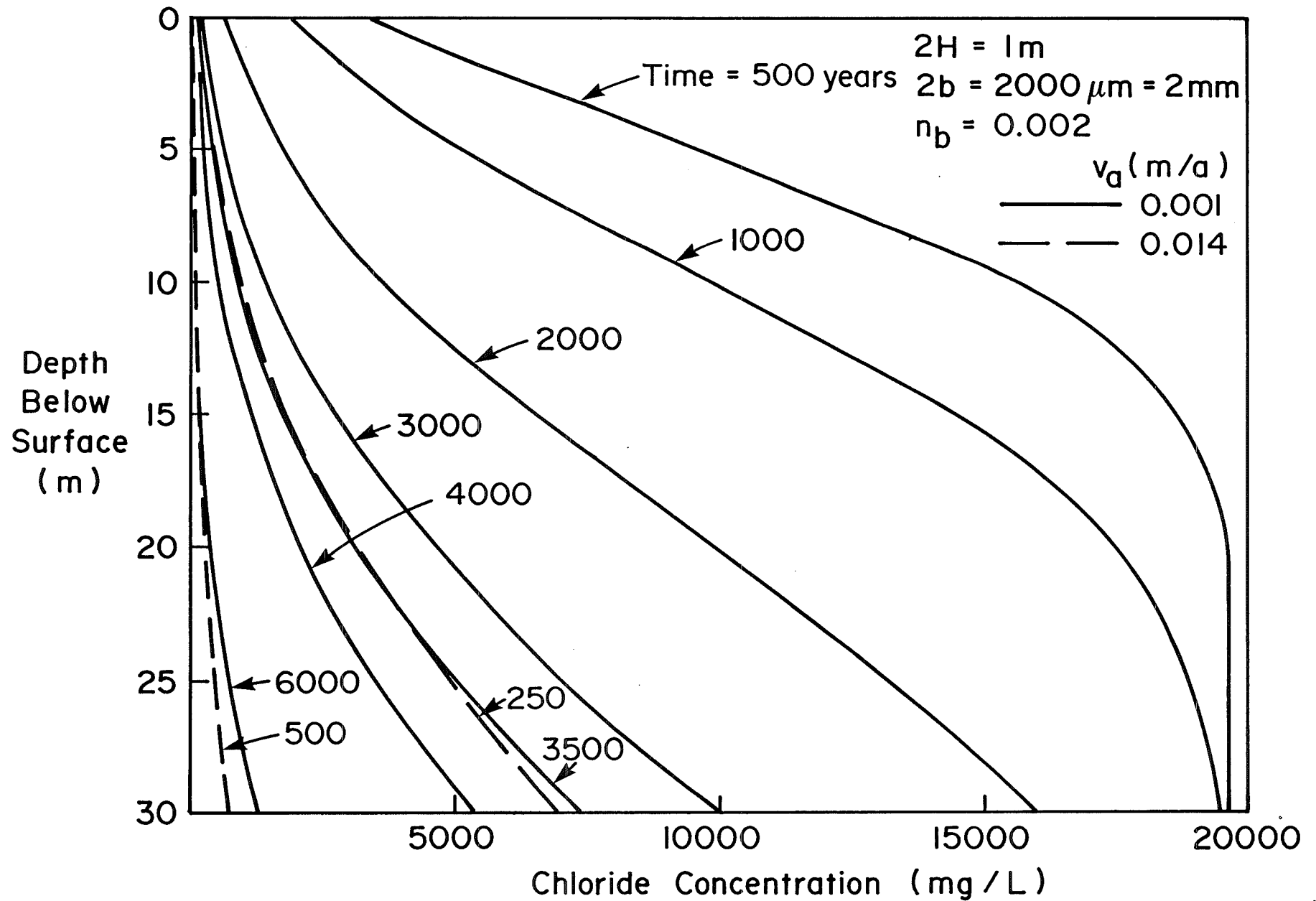


FIGURE 17 CALCULATED VARIATION IN CHLORIDE CONCENTRATION WITH DEPTH BASED ON LEACHING OF AN INITIALLY UNIFORM PROFILE BY DOWNWARD FLOW

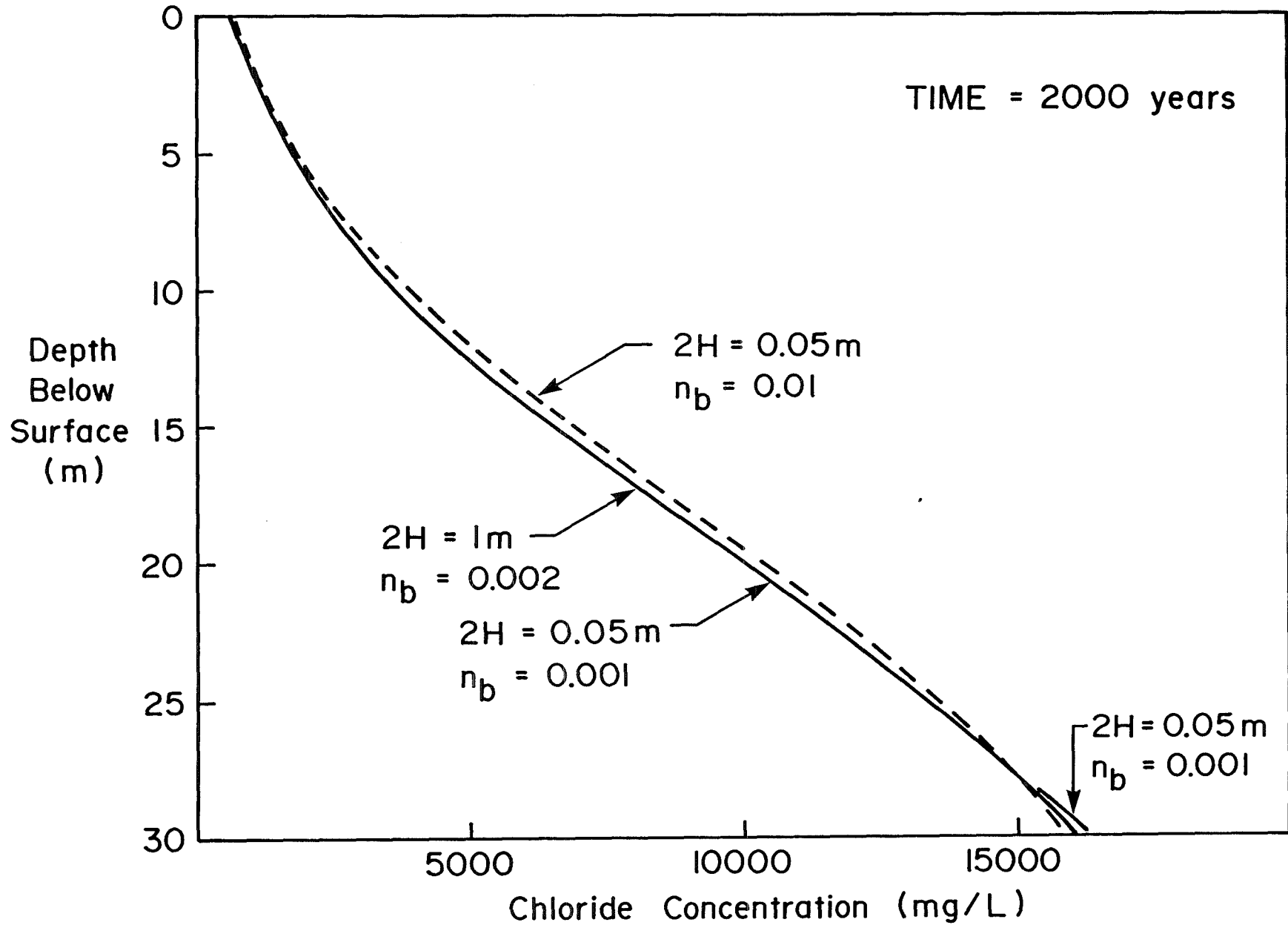


FIGURE 18 EFFECT OF FRACTURE SPACING AND POROSITY ON CALCULATED CONCENTRATION PROFILE
 $v_a = 0.001 \text{ m/a}$