

# **The Predictive Assessment of the Migration of Leachate in the Subsoils Surrounding Mine Tailing and Dredging Spoil Sites**

M. LOXHAM<sup>1</sup>

## **1 Introduction**

Mine tailings and the very similar dredging spoil form just one example of a wide category of bulk inorganic wastes that can be disposed of by surface storage or landfilling. Other wastes include pulverised fuel ash, waste phosphate gypsum, cement fly ash, desulphurisation sludges and domestic waste incinerator ash. The disposal can be simply passive or have a geotechnical function such as land make-up or embankment fill.

This category of wastes differs from conventional chemical waste primarily in its volume and stability and in the fact that the largest proportion of the waste is in no way hazardous and differs only marginally, if at all, from naturally occurring minerals.

Unfortunately, they do contain trace levels of leachable toxics such as heavy metals, amphoteric, metalloids, fluorides and in the case of dredging spoil and much incinerator ash, persistent organic residues. In some cases radionuclides can be released such as radium, the actinides or the gas radon. The concentrations in the leachate are invariably low but none the less environmentally significant. Furthermore, the release can take place over very long time spans and potentially pollute large areas.

This long-term impact is to be distinguished from short-term impact by the displacement of the associated pore water or transport water. The pore water very often contains comparatively high levels of processing fluids such as acids, organics or bases as well as soluble non-toxic salts. In fact, many of these wastes are characterised by a short-term peak release from these fluids followed by a long-term steady state one, by ordinary leaching processes.

This release of toxic material can significantly degrade the ground- and surface waters in the surrounding environment. It is usually feasible and cost effective to control the releases to surface waters and some measures can be taken against the short-term release into the groundwater. However, the long-term threat to the groundwater is more difficult to address.

The most significant mechanism leading to the spread of the toxic components from the immediate vicinity of the site into the environment is that of advection with the groundwater. The quantitative assessment of this is the key to the overall safety analysis of many disposal sites and to the rational, cost-effective design of any necessary environmental protection countermeasures.

---

<sup>1</sup>Delft Soil Mechanics Laboratory, P.O. 69, 2600 AB Delft, The Netherlands

In recent years great strides have been made in predictive advection modelling techniques and this is fortunate as the impact time factors for both the dispersion of the toxics and any countermeasures taken against it, is so long that direct experimental simulation is out of the question. Typically, environmental engineers are required to assess the effects of leachate migration on the quality of the surrounding aquifers on time scales of up to thousands of years.

In the following, a description will be given of a practical methodology for the quantification of these advection effects without going into the mathematical physics behind them. For these, the reader is referred to standard texts on groundwater pollution (Bear 1972; Fried 1975; Kirkham and Powers 1972). The methodology has been found useful and adequate for a wide variety of problems and some examples will be given. Emphasis is placed on the optimisation of investigatory effort to reach a given conclusion and in particular that of keeping to a minimum the highly expensive field and laboratory work.

## 2 Source-Path-Target Methodology

Before going into computational details it is worthwhile to consider the overall methodology of site assessment. In almost all cases it is possible to identify three elements in the safety assessment problem. These are:

1. A source;
2. A target;
3. A pathway connecting them.

This is illustrated in Fig 1.

In practice multiple targets and associated pathways can be identified and the source-path-target diagrams can become very complicated.

The source is characterised by its nature and its emission strength. This can be an advective leachate flux and concentration or if this flux is very small, a diffusive flux. In general, the emission will change with both time and spatially over the source site. The emission flux is often referred to as the “aquifer loading”. This reflects the convention that the unsaturated zone under the site down to the groundwater table and any artificial barriers such as liners under the site are included as part of the source description rather than that of the path. The release scenarios considered are more often than not the result of risk analysis studies of the (composite) source.

The target is characterised by a maximum allowable impact value. Typical targets are streams, abstraction wells and other outflows. However, more diffuse targets can be imagined such as a crop-rooting zone, a nature reserve or even a whole aquifer. In practice the target is specified as a maximum allowable concentration value (MAC value) and the actual numerical value of these are the subject of much controversy. Drinking water standards are often specified and the regulatory authorities usually do not allow credit to be taken for in-target dilution effects as for example the mixing of an outflow into the bulk water flow of a river.

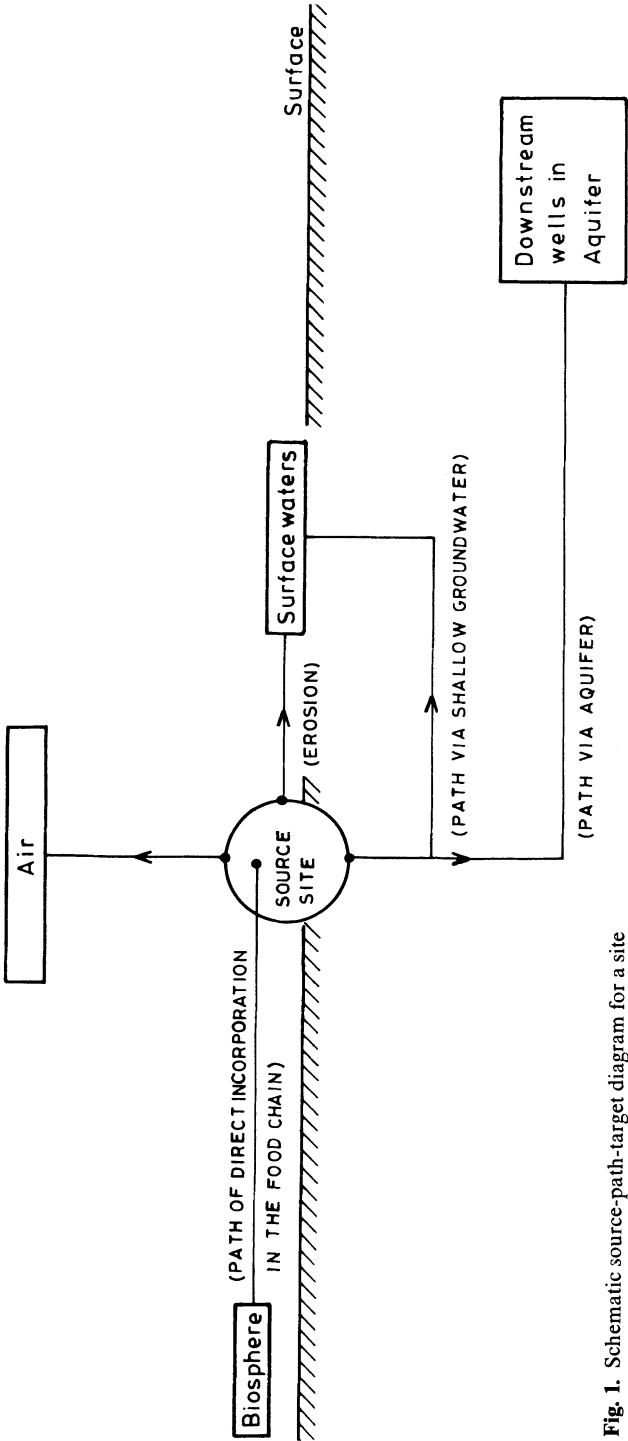


Fig. 1. Schematic source-path-target diagram for a site

The source and target are connected along a path. This path is characterised by its dynamics. Typically, the concentrations decrease along the path by dispersion and dilution effects, but also by radioactive or biological decay. The quantification of the phenomena along the path is the objective of most site-assessment studies.

If the expected impact of a particular site at the target is found to be unacceptable, then countermeasures have to be considered. These can be directed at any one of the three components of the S-P-T system:

1. The source can be reduced (or removed);
2. The target can be moved;
3. The path can be manipulated.

An example of this manipulation is illustrated in Fig. 2 for the case of a cut-off wall backed up by a guard well.

Figure 2 also illustrates two important features of the engineering approach to the problem. Firstly, the difference between the MAC value and the predicted concentration at the target can be taken as a measure of the impact safety factor being carried in the design. This factor is usually specified from both economic and other non-technical considerations. This value has a range of uncertainty. This can be illustrated by considering Fig. 3, where the overall engineering costs are set out against the degree of target protection required.

There is in fact an envelope of design curves for any given problem depending upon the uncertainty accepted in the final system performance and the thereby implied safety factors. The second aspect to be noted is that all engineering

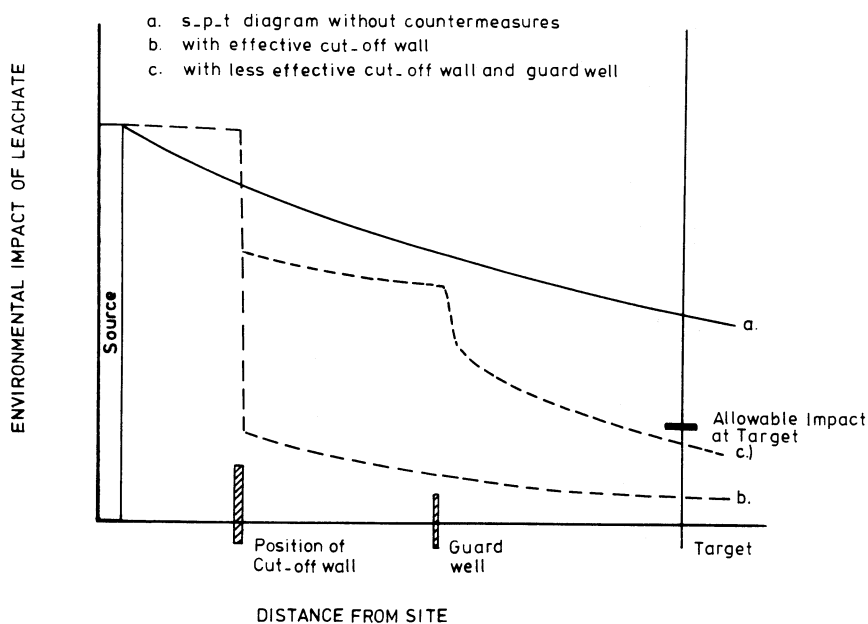
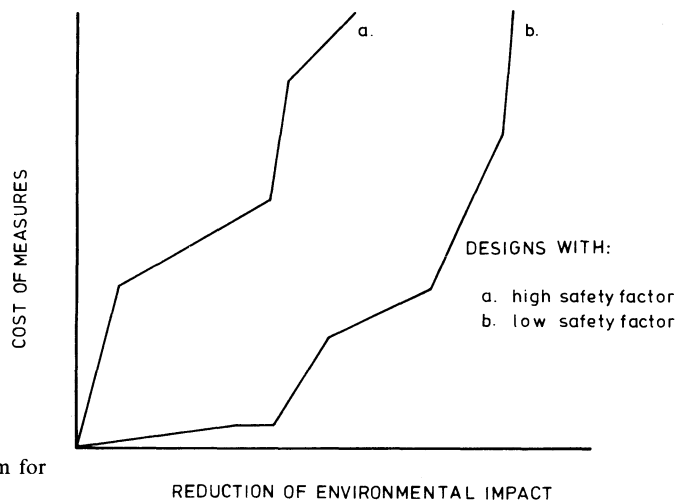


Fig. 2. Effect of countermeasures on s-p-t diagram



**Fig. 3.** Cost effect diagram for countermeasures

solutions to environmental problems must also be considered in their failure mode and must be subjected to a full risk analysis study. In the hypothetical case shown in Fig. 2 this has led to the back-up guard well being sunk in order to provide a minimum barrier that could be activated if the cut-off wall were to fail.

It is the environmental engineer's task to generate analogous information to that summarised in Figs. 1 through 3 for particular sites and scenarios.

### 3 Groundwater Flow and Contaminant Migration

The general methodology for contaminant migration studies is as follows:

1. Measure or calculate the regional groundwater flow;
2. Measure the regional and local groundwater qualities;
3. Measure or model the groundwater flow in the local region either at the existing site or its future location;
4. Identify the appropriate targets;
5. Specify the operation scenarios for the site;
6. Define the relevant source terms;
7. Estimate the migration parameters;
8. Model the migration patterns using the contaminant migration models;
9. Conduct a sensitivity analysis on results from the point of view of the parameter choices and levels;
10. If necessary, reiterate (7) through (10) until the desired precision is achieved.

In this section the emphasis will be placed on the techniques needed to conduct step (8) as efficiently as possible.

The groundwater flow between the source and target, usually referred to as the local flow pattern, is distinct from that of the rest of the watershed, the so-called

regional pattern. In the green-field situation before the site is in place, the local system is usually controlled by the regional one and can be established by conventional geohydrological techniques (see for example Rushton and Redshaw 1975; Wen-Hsiung Li 1972).

Similar considerations apply for the chemistry of the local and regional systems. This so-called background or zero concentration profile is usually established by groundwater sampling and chemical analysis. A priori calculation is not possible.

The construction and start-up phase of a site is commonly accompanied by major disruptions of the local system. It is possible that pumping and transport water are drained from the site to the local system and surface watercourses can be radically altered. Pore water resulting from waste consolidation can also drain to the local system and there can be major infiltration changes as the balance between rainfall and evapotranspiration in the region of the site is disturbed. Finally, tailings dams can result in ponded conditions over large areas. This initial pollution impact, albeit temporary, can have serious consequences for the overall aquifer quality, introducing a slug of contaminant that slowly moves downstream, polluting wells and rivers long after site construction has finished.

For a site overlying a salt water aquifer there is an additional complication, i.e. the effective total infiltration fluxes of contaminated but comparatively fresh water can lead to the development of a freshwater lens under the site with the consequent radical changes in groundwater flow patterns.

After operation has finished, in the caretaking stages and beyond, the flow pattern settles down again to a steady state that often closely resembles the original green-field situation.

There are two exceptions to this. Firstly, in the case where major engineering elements have been introduced into the aquifer such as cut-off walls or grout barriers and secondly, where the tailings themselves intrude into the aquifer. An example of this is where dredging spoil, which has a final permeability in the order of that of clay, is landfilled and forms a large plug in the aquifer around which the groundwater has to flow.

In all the above cases the groundwater flow pattern can be followed by computer modelling techniques. Large, three-dimensional finite element or finite difference groundwater modelling codes are widely available today and will not be discussed further except to point out that there are significant cost savings to be made by calculating the sequence of operating stages of the site as a series of hydraulic steady states rather than trying to follow the exact time dependence of the system. Typically, the hydraulic response time of the system is much shorter than the operating scenario time scales and the above simplification is usually justified. An exception is where major changes in the groundwater table are introduced.

Once the groundwater flow pattern has been established it is possible to calculate the developing contaminant distribution pattern by using the conventional convection – dispersion equation:

$$\frac{\delta c}{\delta t} + \frac{\delta s}{\delta t} = -\nabla(v c) + \nabla D \cdot \nabla C - \lambda c, \quad (1)$$

where  $C$  and  $S$  are the dissolved and adsorbed concentrations respectively,  $t$  the time,  $D$  the hydrodynamic dispersion tensor and  $v$  the (mean local) in-pore water velocity vector.  $\lambda$  is the (first order) decay term for radioactive decay or biological effects.

Although in no way necessary for modern computer implementation of this model, it will be convenient for our purposes to rewrite the adsorption term in terms of the conventional retardation factor  $R$ , where:

$$R = (1 + K), \quad (2)$$

and  $K$  is the distribution coefficient given by:

$$K = s/c, \quad (3)$$

where  $S$  is expressed as mass adsorbed per unit volume associated pore fluid. This results in:

$$\frac{\delta c}{\delta t} = \frac{1}{R} \left\{ -\nabla(vc) + \nabla D \cdot \nabla c - \lambda c \right\}, \quad (4)$$

which has to be solved once the spatial distribution of the parameters, the initial and the boundary conditions, are specified.

The initial conditions can be obtained from the zero situation measurements of the spatial distribution of the concentrations in the area to be studied. Often it is assumed that these are zero and that for calculation purposes the emission from the site is considered in terms of incremental load. This can be an erroneous procedure especially in areas where mine tailings are dumped near the mining activity itself and this approach can lead to large overestimates of the environmental impact of the dump.

The mathematical structure of the convection – dispersion equation requires that two boundary conditions be specified. The most common formulation is to specify the input flux as a function of time at the site and the output fluxes at the calculation domain boundaries. A less useful combination is that of concentrations at the site and domain boundaries. The input fluxes at the site are themselves usually the results of modelling studies of the hydrology and speciation chemistry applied to the waste itself.

Before discussing possible solutions to this general equation, it is worthwhile to consider it in more detail. Any solution to the equation requires input values for the water velocity (or as this equation is usually solved in combination with the Laplace formulation of Darcy's law, the hydraulic conductivities), the dispersion coefficients, the adsorption coefficients and any decay rate coefficients. These have to be specified for both space and time over the calculation domain.

These parameters can be measured in the field, in the laboratory, estimated from the nature of the ground conditions, assumed from experience or common sense, or just simply guessed. Adequate parameter estimation is the major difficulty in the effective use of predictive modelling techniques.

The parameters usually exhibit a wide variation even for rather simple geohydrological settings. Typically, the hydraulic conductivity and the adsorption coefficient can vary several orders of magnitude on a 100-m-length scale.

Fortunately, the objective of the calculation within the context of an overall safety analysis demands ranges of the results rather than specific exact values and techniques based on sensitivity analysis and stochastic approaches can be used. The major cost and time factors in the assessment study are associated with the parameter estimation. The sensitivity analysis strategy is directed to minimising these.

Usually a calculation is made using preliminary information and the results subjected to the parameter sensitivity analysis. Four possible result/parameter combinations can be expected. There are parameters which are well defined and of importance in deciding the overall outcome of the safety analysis, and parameters that are either well or poorly defined but in any case irrelevant. These three combinations do not impact on the next iteration of the parameter estimation process. However, there is a last category, i.e. the parameter is of importance but poorly defined and this has to be remeasured. The calculation is then repeated with the new estimate and the sensitivity analysis reiterated, and so on and so forth until the required precision and policy discrimination in the final overall analysis is achieved.

In practice one or at the most two iterations will be permitted and perfect answers can never be expected. This automatically implies that usually massive safety factors have to be assumed in the analysis and this alone puts a limit on the degree of effort and modelling complexity that can be usefully employed.

Examination of the equation shows that it is the velocity (or hydraulic conductivity) that controls the downstream spread of the contaminant; the dispersion coefficient controls the lateral spread and the form of the downstream contaminant front; and the adsorption coefficient the time scale within which this occurs. The adsorption could, however, delay the contaminant long enough for the decay processes to make significant inroads into the contaminant inventory on the migration paths, thereby reducing further the impact at the target.

In the following sections some consideration will be given to specific solutions of the above equation.

## **4 A Site in an Infinite Homogeneous Aquifer**

In this section the case is considered of a site with an emission into a large aquifer of constant properties. The objective of the calculation will be to define the concentrations expected downstream from the site. Whilst the physical setting of the problem is highly simplified, it is none the less complicated enough to illustrate the major features of the problem which are to be found in more difficult cases. Furthermore, the simple analysis given here is usually the one chosen to give the first estimate of the nature of the problem and to help focus the major computer simulation, which is normally required at the final stage.

Consider the situation illustrated in Fig. 4 which shows a site above an aquifer.

The groundwater velocity in the aquifer is constant in both space and time. The aquifer is confined and has constant thickness, dispersion and adsorption coefficients. At the site the confining barrier clay has been excavated and replaced

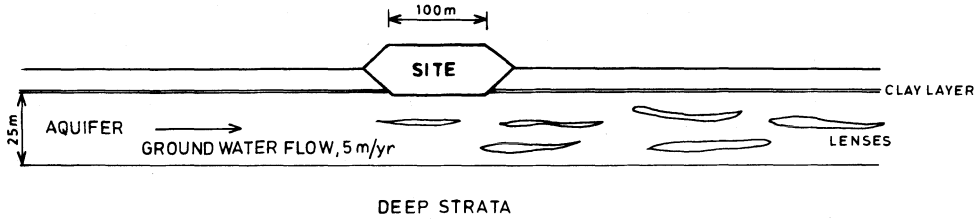


Fig. 4. Site in its surroundings

by tailings, allowing direct penetration of leachate into the aquifer (this occurs more frequently than one would hope or imagine!).

There is a flux of contaminant from the site into the aquifer. For the moment this will be considered a constant but this is in no way essential. The contaminant flux does not alter the water flow pattern under the aquifer.

These conditions have been chosen so that the convection dispersion equation has a simple analytical solution that can easily be programmed for a home or microcomputer. Thus (Wen-Hsiung Li 1972):

$$c(x,y,z,t) = \frac{M}{8\pi D r} \exp\left(\frac{vx}{2D}\right) \left\{ \exp\left(\frac{vr}{2D}\right) \left( 1 - \operatorname{erf} \left[ \frac{\frac{vt}{r} + r}{\sqrt{4Dt/R}} \right] \right) + \exp\left(\frac{-vr}{2D}\right) \left( 1 + \operatorname{erf} \left[ \frac{\frac{vt}{R} - r}{\sqrt{4Dt/R}} \right] \right) \right\} \quad (5)$$

and  $r^2 = (x^2 + y^2 + z^2)$ ,

where  $M$  is the specific flux from the site to the aquifer ( $t^{-1}$ ) and the other parameters have been defined earlier. The solution given here is for a point source, that is where the calculations are to be made for a location at some distance from the site, but can be easily extended to finite or multiple sources by summation and for time-variable sources by convolution techniques (Courant and Hilbert 1967).

For the case of a not too thick aquifer, Eq. (5) can be reduced to an even simpler form:

$$c(x,y,z) = \frac{m}{4\pi D n} \int_0^t \frac{1}{\tau} \exp\left[ -\frac{\left(x - \frac{v\tau}{R}\right)^2 - y^2}{4D\tau/R} \right] d\tau, \quad (6)$$

where  $m$  is the specific line flux ( $t \text{ s}^{-1} \text{ m}^{-1}$  aquifer thickness).

In practice Eq. (5) is chosen for cases where the bottom of the aquifer is not defined, such as for sites above fractured rock or karst formations. In this case care has to be taken in the interpretation of the definition of concentration. The concentration measured in the field ( $C_m$ ) is the average over the depth of the sampling filter and is related to the point calculated concentration by:

$$c_m = \frac{1}{(h_1 - h_2)} \int_{h_2}^{h_1} c(x,y,h) dh, \quad (7)$$

and this should be taken into account where necessary. For many cases where the site is above a sedimentary aquifer, it is possible to use Eq. (6) for estimating purposes. Whilst the question as to at which point Eq. (5) has to be used rather than Eq. (6) is interesting; in practice, it is largely irrelevant. This is because the overall problem itself becomes much too complicated for this simple analysis in the immediate region of the site and recourse has to be made anyway to proper finite difference representations of the convection dispersion equation itself.

It is now possible to use these equations to illustrate some important features of the problem. Firstly, it is to be noted that they predict the contaminant plume to be ellipsoid in form with the maximum concentration on the downstream centre line. This is shown in Fig. 5.

A second general feature is that these equations have a pseudo-steady state behaviour at long times. Explicitly, Eq. (6) reduces to (Bear 1975):

$$c(x,y,\infty) = \frac{m}{2\pi D} \cdot \exp\left(\frac{vx}{2D}\right) \cdot K_0\left[\frac{v}{2D} \sqrt{x^2 + y^2}\right] \quad (8)$$

where  $K_0$  is the modified Bessel function of the second kind and zero order. The downstream concentrations develop to this steady state value asymptotically but in practice the value is reached at a time of about  $(4 \cdot R \cdot v / x)$ . In other words, there is a finite extent of the pollution plume for any given concentration level and parameter combination. The development towards this steady state is shown in Fig. 6 for a typical combination of parameters.

It should be noted that the retardation coefficient,  $R$ , does not occur in Eq. (8) and the maximum plume volume is not influenced by adsorption, only the time taken to reach it.

In order to gain some insight into the sort of numerical values to be expected, consider a typical case where a fly-ash site is releasing a fluoride anion at a total flux of  $150 \text{ kg yr}^{-1}$  into an aquifer of 25-m thickness with an in-pore water velocity of  $5 \text{ m yr}^{-1}$  (Fig. 4). In Fig. 7 the contaminant plume extent at the drinking water concentration of 1 ppm is illustrated as a function of the dispersion coefficient.

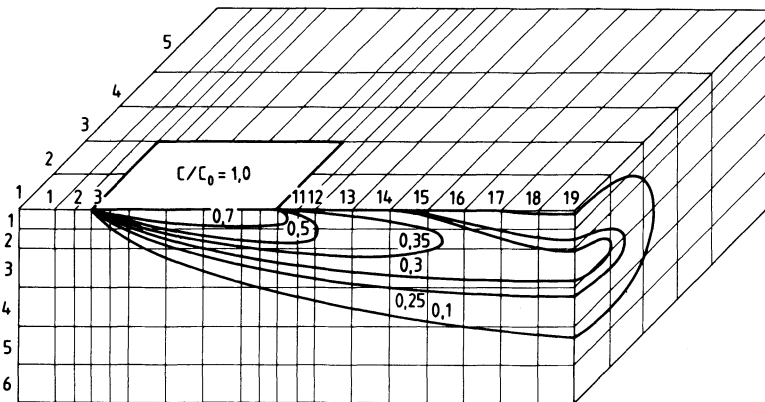


Fig. 5. Three-dimensional concentration development

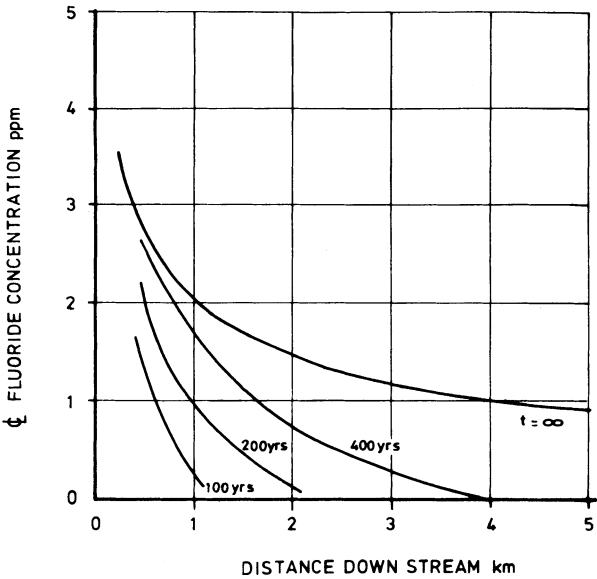


Fig. 6. Concentration buildup to steady state valve

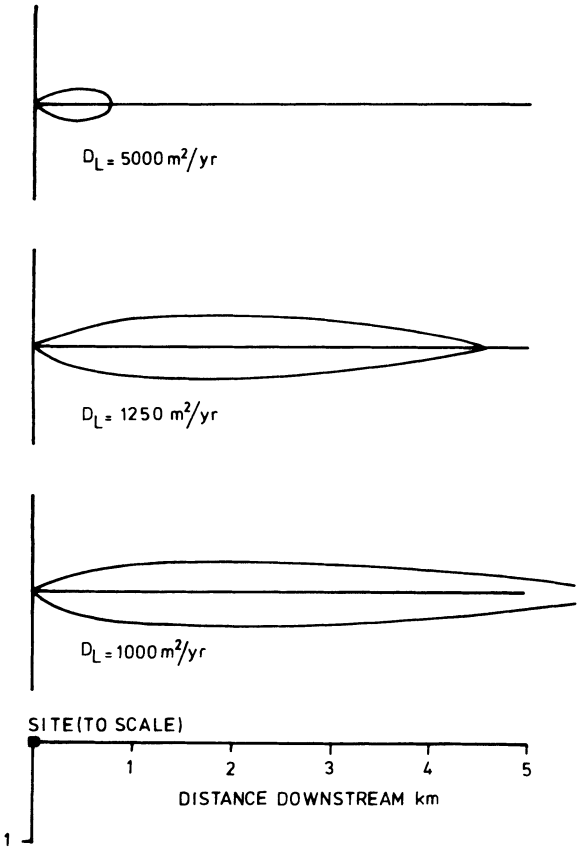


Fig. 7. The downstream plume

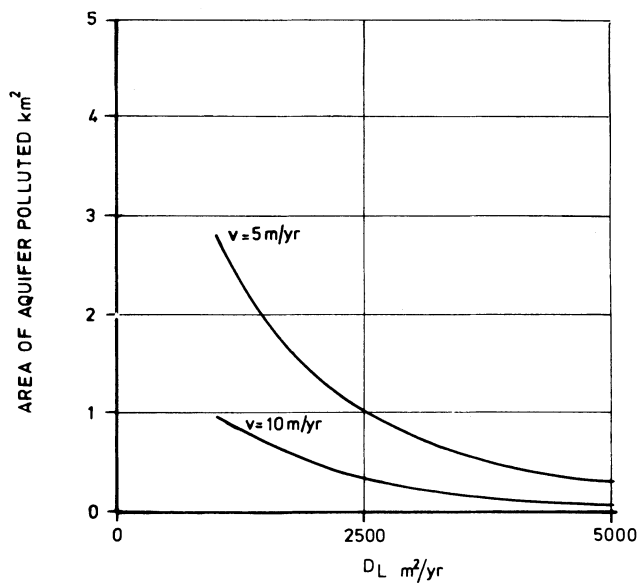


Fig. 8. Area of aquifer polluted

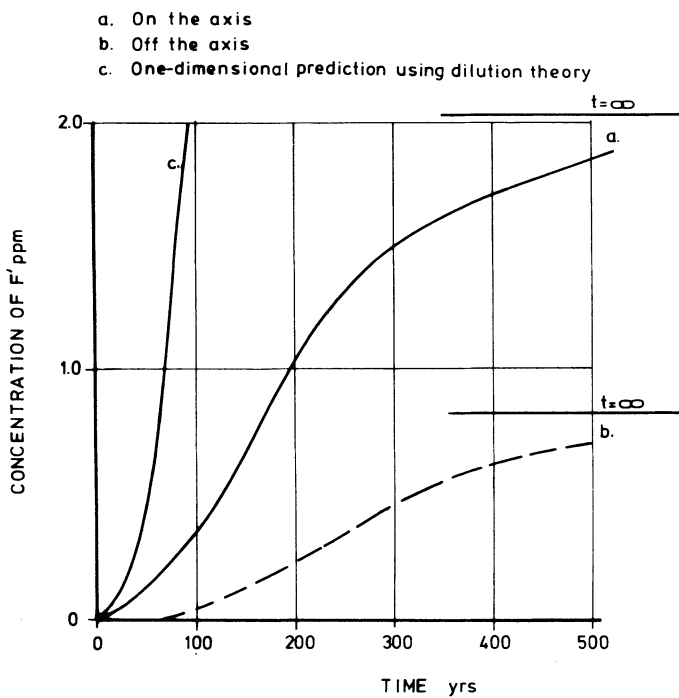


Fig. 9. Breakthrough curves 1 km downstream

As can be seen in Fig. 7, the effect of decreasing the value of the dispersion coefficient is to cause a longer plume downstream. With high dispersion behaviour the contamination is quickly washed out around the site. The area of aquifer contaminated with water that has a concentration above the MAC value is shown in Fig. 8 as a function of both velocity and the dispersion coefficient.

Finally, and again as an illustration, the concentration build-up curve at a point downstream is shown in Fig. 9.

This can be compared to that obtained from a one-dimensional simulation using the “dilution factor” source concentration (emission per unit width of a site normal to groundwater flow/groundwater flux). This is also shown in Fig. 9 and illustrates that on the flow axis the one-dimensional calculation gives a large overestimate of the concentration development. There is a considerable improvement by treating the problem as a two-dimensional one.

Figure 9 also illustrates another important feature of the breakthrough of a contaminant at the target and that is the so-called toe effect. This is the small but often environmentally significant concentration buildup ahead of the main arrival time front. The extent of this early breakthrough is largely controlled by the dispersion coefficient reflecting the heterogeneous nature of the system. In many problems it is the arrival time of the toe that is significant and not that of the main front.

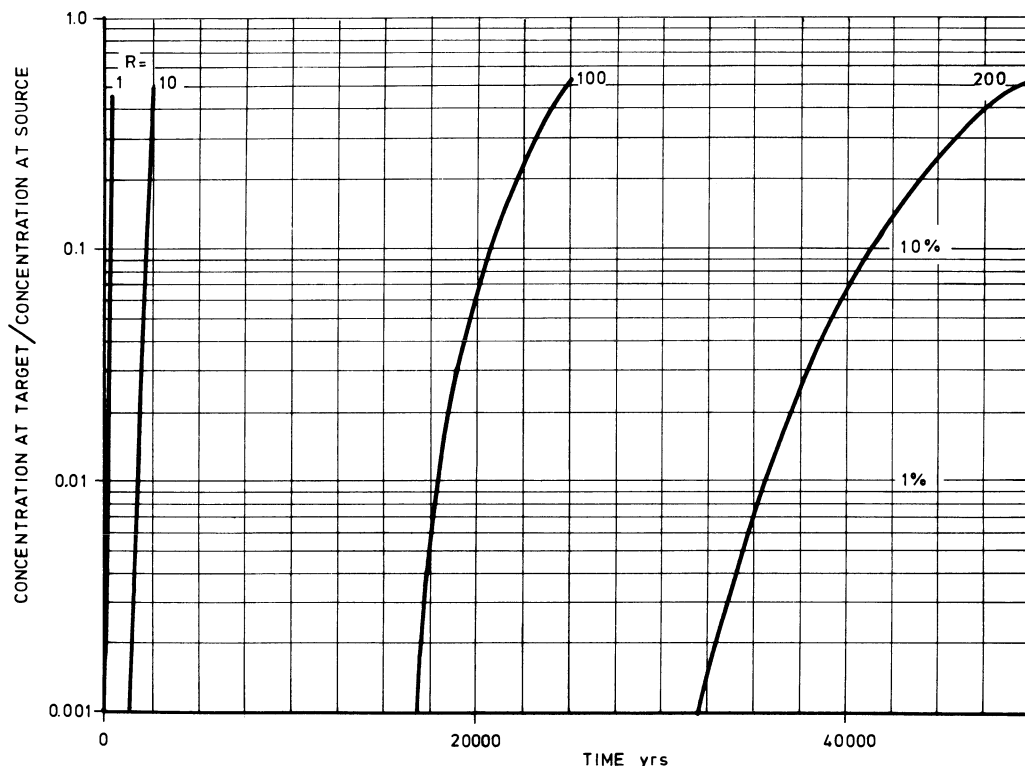
## 5 Adsorption and the Problem of Real Systems

In the previous section we have considered the case of a non-adsorbing component and the maximum plume size that can be expected. It has been shown that there is a steady state maximum size of this plume that is reached typically in thousands of years. In this section the effect of adsorption of the migrating components on the development time of the plume will be examined in more detail.

These effects can be of dominating importance in the safety analysis. The retardation coefficients as given by Eq. (3) can vary from  $< 5$  for sandy systems to more than 10,000 for heavy metals on clays and peats. Extensive reviews of adsorption measurements on a laboratory scale have been published (see for example Bolt and Bruggenwert 1978).

In general terms, soil organic material, clay surfaces and precipitated iron and aluminium oxides are responsible for adsorption of heavy metals and polar organics. Non-polar organics and anions are much more poorly adsorbed. Adsorption on the active surfaces of soil components should be distinguished from precipitation from the pore solution itself. However, in many operational procedures to determine the adsorption coefficient these phenomena are difficult to separate and are often lumped together in the presentation of the results.

Unfortunately, there is little or no field adsorption data available at this time. Furthermore, the actual adsorption behaviour is controlled by a wide range of physical and chemical factors such as the state of the adsorbant surface, pH, redox potential, organic matter content and component speciation. These factors cannot only change in time, but also under the influence of the pore water chemistry



**Fig. 10.** Effect of retardation concentration development

changes associated with the migration front itself. Finally, it is not uncommon for different laboratories to report adsorption coefficients that differ by two or three orders of magnitude for nominally identical systems.

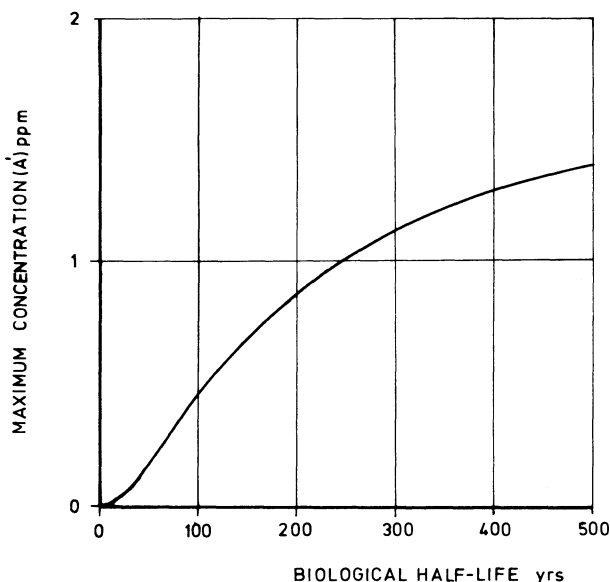
The influence of different adsorption coefficients on the breakthrough behaviour for the example chosen above (cf. Fig. 9) is shown in Fig. 10.

The effect of a high adsorption coefficient can be to delay the breakthrough until essentially geological time scales. This is true for both the main front and the toe of the breakthrough.

The adsorption can be seen both as a barrier slowing down the migration front and as a buffer resulting in the storage of toxic material leaching out of the site in the surrounding soils and thus preventing its convection to the target. It is, however, only rarely that the total adsorption capacity of the soils on the path to the target is so large that all the site inventory can be held back in this way.

For a component undergoing radioactive or biological decay the adsorption barrier can delay the migration effectively enough to ensure that none of the migrating component reaches the target. This is illustrated in Fig. 11. This is the basis of adsorption barrier technology in radioactive waste isolation.

Usually the quoted values of the adsorption coefficients are obtained from laboratory experiments on disturbed samples and give maximum values.



**Fig. 11.** Breakthrough curve for an organic component (A) at 1 km downstream assuming different half-lives

The major difficulty in assessing the field effective value of the adsorption barrier lies, however, in the heterogeneous nature of real soils and aquifers. Typically, the permeabilities and adsorption coefficients can vary over several orders of magnitude on a 100 m-length scale in a sedimentary environment.

The adsorbing soil components such as clays tend to occur as finite lenses, reflecting their sedimentary origins. These lenses have generally a low hydraulic conductivity so that the water (and contaminant) tends to flow around rather than through them. In this way it is possible that some contaminants that would otherwise have been retarded, can by-pass the adsorption sites and break through earlier than expected. This effect is seen most clearly in the toe of the breakthrough curve.

The actual mobilisation of the potential adsorption capacity is a function of several factors such as the actual geometry of the system, the lens properties and the time scales of the flow regime. This last dependency arises because for clay lenses the hydraulic conductivity is so low that the adsorption sites can only be accessed by diffusion mechanisms and these operate independently of the surrounding water velocity.

In practice two extremes can be identified, one where all the adsorption sites can be accessed and one where only molecular diffusion access is possible. The difference between these two extremes is shown schematically in Fig. 12.

In order to assess the availability of adsorption capacity in lens systems, calculations were made (Loxham 1983; van Meurs et al. 1985) on predefined heterogeneous systems using one of the large, modern, finite-difference, contaminant-migration codes (DSML 1982). The calculations showed trends similar to those given in Fig. 12, including the fast breakthrough at low initial concentrations.

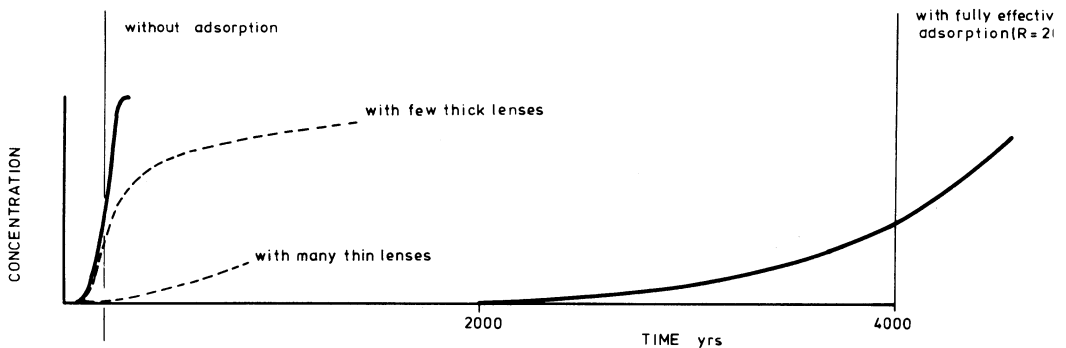


Fig. 12. Schematic effect of concentrating the adsorption capacity in fewer thicker lenses

In practice the amount of profile information required to perform such calculations will never be available. The question then arises as to what is the minimum of field information required to support a given conclusion.

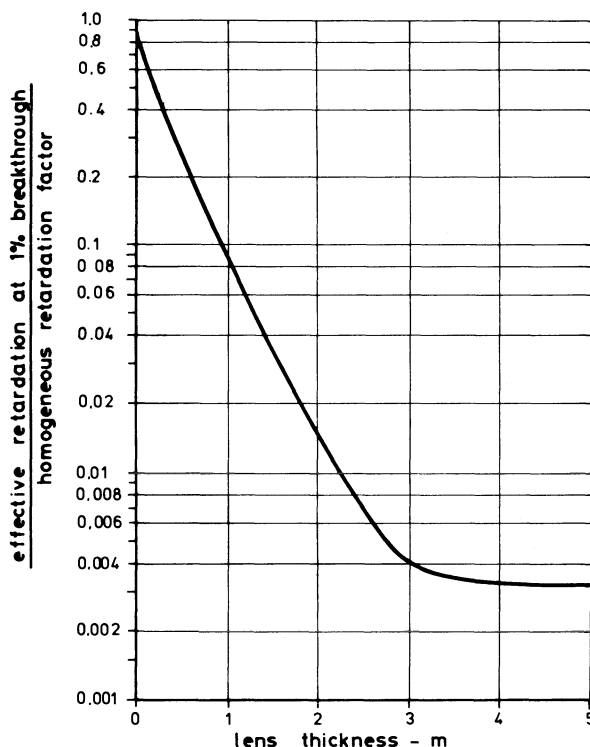
The field information required concerns the actual position, or at least the thicknesses and stratigraphic successions of the lenses. These can be determined from borehole logs and geophysical techniques.

Three levels of information can be identified:

1. Overall mean soil composition without structure differentiation;
2. Stratigraphic succession with layer thickness data but without spatial distributions;
3. Complete profile information.

Studies (Loxham 1983) have been made on the utility of the information at each of these levels and the level of modelling sophistication appropriate to the field data base for landfilling low level radioactive wastes. The conclusions of these studies have general applicability to this sort of contaminant safety assessment. It was concluded that contaminant modelling with the last level of information was superior to the simpler systems, but involved a prohibitive investment in field work. Models using only first level information generally gave a large overestimate of the value of the adsorption barrier and correspondingly low (sometimes fractional!) overall safety factors in the site design. Surprisingly, calculations, using level two information, where the lens thicknesses were available but not their positions relative to each other, gave quite reasonable results.

In Fig. 13 one of the results of the study is illustrated. In this figure the apparent retardation coefficient found from the fully heterogeneous calculation divided by the maximum value is used as a measure of the mobilisation of the adsorption barrier. It can be seen that for very thin lenses all the adsorption capacity can be taken into account and for very thick lenses it is better to discount the adsorption barrier completely. Similar calculations can be performed for other sites and stratas but the results show the same trends. For lenses of intermediate thickness simple but adequate estimates can be made using dual porosity models (Skopp and Warrick 1974; Loxham and van Meurs 1984).



**Fig. 13.** Influence of lens thickness on effective retardation factor for clay lenses

This study suggests that it is possible to tailor the field work down to determining the stratigraphic data of the soils on the paths and obtaining undisturbed samples of representative strata for laboratory determination of the adsorption behaviour.

Finally, it should be noted that for fracture rock systems and karsts, a similar problem occurs in that the bulk of the rock has a volumetric holding capacity for contaminants that can only be accessed by diffusion from the hydraulically active fissures. However, the systems do differ in that in fissure systems the adsorption capacity proper tends to be concentrated at the surface of the fissure or in microcracks in its vicinity (Neretnieks 1980; Tang et al. 1981; Sudicky and Frind 1982).

## 6 Engineered Pollution Control Options

It was pointed out earlier that remedial actions for the case where the environmental impacts of the site on the quality of the aquifer are unacceptable can be directed towards either the source or pathway. Source-orientated measures can include prevention of leaching by excluding infiltration, stabilisation of the tailings by chemical means or as a last resort excavation and removal (which presents a

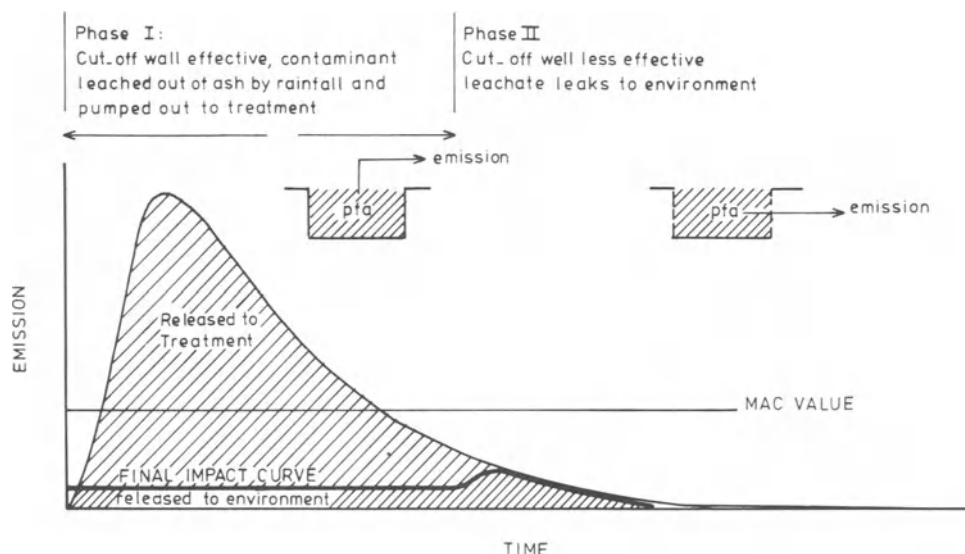
problem elsewhere). These methods will not be considered here and attention will be directed to pathway control, such as cut-off walls, grout barriers, guard wells, etc.

These methods can be very effective and have found wide application. However, one general feature of all engineering options for path control is that they are relatively short-lived measures and cannot be expected to guarantee, on their own, long-term environmental safety. In other words, they should only be considered where the passive long-term behaviour of the site tends towards one of acceptable environmental impacts.

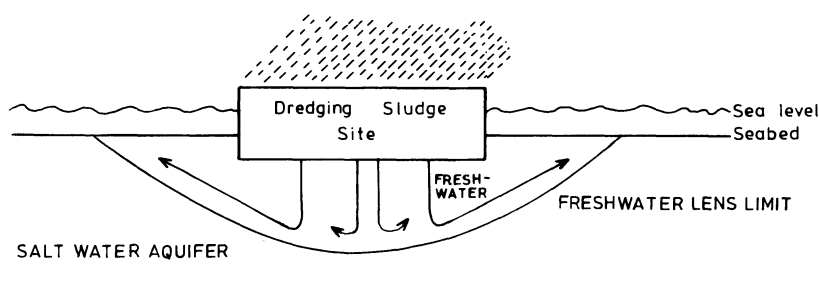
An example of this is the landfilling of PFA behind a deep cut-off wall. In the short term the wall protects the aquifers in the surroundings from the initial high leachate peak. This is washed out of the ash in a period of some 50 years by infiltrating rainfall which, in turn, is pumped up from within the site and treated. In the long term, when it is to be expected that the cut-off wall might fail, the residual impact of the remaining leachate will be acceptable for the surroundings. This strategy is illustrated in Fig. 14.

A similar, "fail-safe" strategy has been proposed for an offshore harbour, dredging-spoil dump site (van Meurs et al. 1985) in the Netherlands. In this case the initial peak leachate load is caused by consolidation water being pressed out into the underlying strata and by the initial chemistry of the spoil. This consolidation water is, however, less dense than the surrounding seawater and forms a freshwater lens under the site. The fluid dynamics of this lens are such that contaminant is substantially prevented from migrating into the surrounding aquifers as illustrated in Fig. 15 and this saves the considerable costs of a deep cut-off wall.

In the long term the spoil consolidates to give a monolith with a very low permeability and correspondingly low pore water emission flux. Whilst this flux is no longer capable of maintaining the favourable barrier function of the freshwater



**Fig. 14.** PFA containment strategy



**Fig. 15.** The action of a fresh (ground) water lens

lens, the associated contaminant emissions are low enough for there to be no long-term pollution problem from the site.

In the past much has been made of guard well techniques to protect aquifers. In principle, the well is sunk and pumped at a rate sufficient to pull all the contaminants into the well and away from the target. Many calculation techniques have been developed to optimise the well placement and pumping rates (van den Akker 1982).

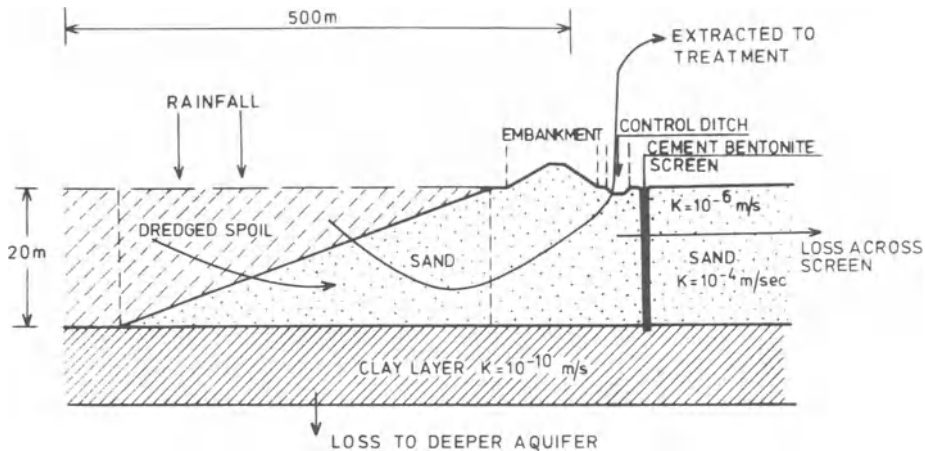
The method has, however, several serious disadvantages. The major one is that the performance of the system depends heavily on the knowledge of the hydraulic conductivity distributions of the aquifer, and these are precisely the quantities subject to the most uncertainty. A second disadvantage is that in practical terms, it would appear that a trade-off has to be made between a few wells pumping a large volume of slightly contaminated water with correspondingly high effluent treatment costs and many small "precision wells" of uncertain performance and very high operating costs. This method has achieved most success in protecting single abstraction points at the target rather than preventing significant emission from a site itself.

In principle, deep cut-off wall isolation technology has the advantage that it uncouples the hydrological regimes within the site from those in the aquifer. Furthermore, construction is possible in a wide range of soil types, also under water. The major disadvantage is that the wall has to be constructed down to an impermeable layer at the bottom of the aquifer so effectively isolating the site (and the groundwater under it) from the rest of the aquifer.

If this strata is absent within economic depths, then the method fails. Attempts to engineer such a horizontal layer by grouting techniques has usually proved either ineffective or prohibitively expensive considering the large areas involved.

A large amount of computer modelling effort has been dedicated to exploring the minimum barrier performance that such a horizontal barrier should have before it starts to significantly impact on the overall safety of the site. This is because in practice the strata actually underlying the sites are either leaky, thin, damaged or all three. Risk analysis techniques are often employed because of the options coupling the site-water management to the leak across the layer.

A typical example of cut-off wall technology is the site at Zelzate in Belgium where the highly contaminated canal dredging spoil is landfilled from the heavily



**Fig. 16.** Migration patterns at Zelzate

industrialised waterway between Gent and the Western Scheldt in the Netherlands (Loxham and Weststrate 1985). A cross-section with the primary contaminant migration patterns is shown in Fig. 16.

The cut-off wall was constructed from cement-bentonite using the panel technique (Xanthakos 1979) down to a thick clay stratum at a depth of 24 m. The wall has a width of 0.8 m. The key to the success of the site is the ring canal shown. This is held on water level control, collecting the initial peak leachate from the tie site for the first 70 years. In this period about 2% of the mobile toxic components are lost via the wall and the underlying clay to the local aquifer and the rest via the ring canal. In due course the quality and quantity of the leachate from the site will rely on the cut-off wall to protect the aquifer, unnecessarily and the basic principle of long-term fail-safe behaviour is satisfied.

## References

- Akker C van den (1982) Numerical analysis of the stream function in plane groundwater flow. Thesis Univ Delft
- Bear J (1972) Dynamics of flow in porous media. Elsevier, New York
- Bear J (1975) Hydraulics of groundwater. Elsevier, Amsterdam
- Bolt GH, Bruggenwert MGM (1978) Soil chemistry. Elsevier, New York
- Courant R, Hilbert D (1967) Methoden der mathematischen Physik Springer, Berlin Heidelberg New York
- DSML (1982) The contaminant transport model VERA. Delft Soil Mech Lab, Delft, The Netherlands
- Fried JJ (1975) Groundwater pollution. Elsevier, Amsterdam
- Kirkham D, Powers WL (1972) Advanced soil physics. Wiley-Interscience, New York
- Loxham M (1983) The influence of soil heterogeneity on the migration of radionuclides in the soil and soil-water systems Final Rep EEC Res Contract 193-81-6-WAS-NL EEC Comm, Brussels
- Loxham M, Meurs GAM van (1984) Int Radioactive waste management Conf BNES, London, p 291
- Loxham M, Weststrate FA (1985) The use of cement-bentonite cut-off wall to contain dredging spoils. Eng Geol 21:359-365

- Meurs GAM van, Loxham M., Weststrate FA (1985) Heterogeneity and its impact on the spread of pollutants. In: Assink JW, Brink WT (eds) Contaminated soil. Nijhoff, Dordrecht, pp 79–87
- Nerethieks I (1980) Diffusion in a rock matrix. *J Geophys Res* 85(88):4379–4397
- Rushton KR, Redshaw SC (1970) Seepage and groundwater flow. John Wiley, New York
- Skopp J, Warrick AM (1974) A two phase model for miscible displacement of reactive solutes in soils. *Soil Soc Am Proc* 38(4):545–550
- Sudicky GA, Frind EO (1982) Contaminant transport in fractured porous media. *Water Resour Res* 18(6):1634–1642
- Tang DH Frind EO, Sudicky GA (1981) Contaminant transport in fractured porous media. *Water Resour Res* 17(3):555–564
- Wen-Hsiung Li (1972) Differential equations of dispersion and groundwater flow. Prentice Hall, New York
- Xanthakos, P (1979) Slurry walls. McGraw Hill, New York