Geoscience Canada

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Volume 23, Number 1, March 1996

URI: https://id.erudit.org/iderudit/geocan23_1ser01

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Publisher(s)

The Geological Association of Canada

ISSN

0315-0941 (print)
1911-4850 (digital)

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Cite this article

Rowe, R. K. & Fraser, M. J. (1996). Engineered Waste Disposal Facilities. *Geoscience Canada*, *23*(1), 1–8.

Article abstract

The selection and design of an engineered waste disposal facility requires consideration of the potential for protection of ground-water quality, predictability of ground-water movement, and potential for disruption of ground-water users. In the design of a waste disposal facility, engineered systems are ofte nincorporated, and the service life of these systems must be considered when assessing their potential impact. The role of modelling in predicting the potential impacts due to the interaction between the hydrogeology and the proposed engineering is discussed, and the potential impact of different landfill designs on ground-water quality is examined.

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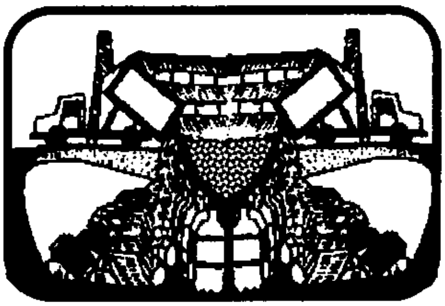

Engineered Waste Disposal Facilities

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SUMMARY
The selection and design of an engineered waste disposal facility requires consideration of the potential for protection of ground-water quality, predictability of ground-water movement, and potential for disruption of ground-water users. In the design of a waste disposal facility, engineered systems are often incorporated, and the service life of these systems must be considered when assessing their potential impact. The role of modelling in predicting the potential impacts due to the interaction between the hydrogeology and the proposed engineering is discussed, and the potential impact of different landfill designs on ground-water quality is examined.

RÉSUMÉ
Dans le choix et la conception d'un ouvrage d'enfouissement sanitaire, on doit considérer les différents moyens de préserver la qualité des eaux souterraines, d'en prédire les mouvements, et évaluer les risques d'en perturber l'utilisation par la population. Il arrive souvent que les projets conçus comportent des ouvrages de génie, et on doit alors estimer la durée de vie de tels ouvrages à l'étape de l'évaluation des répercussions possibles. Dans le présent article, on discute de l'importance de recourir à la modélisation dans la prévision des répercussions possibles découlant de l'interaction de l'ouvrage d'enfouissement sanitaire avec le milieu hydrogéologique, et on étude les répercussions possibles des divers types d'ouvrages d'enfouissement sur la qualité des eaux souterraines.

INTRODUCTION
The selection of a suitable site and design for a waste disposal facility such as a landfill involves the interaction of many disciplines (*e.g.*, geology, geophysics, geochemistry, hydrogeology, geotechnical engineering, and landfill design) in order to characterize a particular site and then develop an appropriate engineered facility for that site. It is necessary to understand the existing site conditions and how the proposed facility will affect existing conditions both in the short term and in the long term. In this context, the potential short-term impacts may extend for up to several decades (*e.g.*, during landfill construction), while the potential long-term impacts may extend over periods of up to several centuries. This latter period of time, during which a landfill will produce contaminants at levels that could have unacceptable impact if they were discharged into the surrounding environment, is often called the contaminating lifespan of the landfill (see also Eyles and Boyce, in press; Birks and Eyles, in press). There are a number of important factors to be considered in the selection of a suitable site, as discussed below.

Potential for Protection of Ground-water Quality
An assessment of the potential for protecting ground-water quality from degradation due to the migration of contaminated water (leachate) from a landfill may involve consideration of natural geological protection, hydraulic protection, and engineered systems.

Natural geological protection generally refers to the ability of a geological feature such as a clay till aquitard to attenuate contaminants as they migrate from the landfill through the aquitard to some potential receptor aquifer (Yanful *et al.*, 1988a, b). This potential for attenuation (*i.e.*, a reduction in concentration of contaminants) will depend on the effective thickness and bulk hydraulic conductivity of the aquitard between the base of the engineered facility and the aquifer. The effective thickness will depend on the existing thickness of the hydrogeological unit (*i.e.*, the aquitard), but also on engineering and other environmental constraints that influence the depth of excavation. Thus, neither the geology/hydrogeology nor the engineered design can be considered in isolation; increasing the depth of excavation may decrease other environmental impacts (*e.g.*, traffic, noise, dust, visual impacts, *etc.*) which affect nearby residents in the short term (which could be decades, as noted above), but this may be traded off against a consequent decrease in natural protection of ground-water quality in the long term. This may then need to be countered by increased engineered protection (discussed later in this paper).

The bulk hydraulic conductivity of an aquitard unit will depend on factors such as density, grain size distribution and mineralogy, and may be controlled by the presence of secondary features such as fractures (*e.g.*, D'Astous *et al.*, 1989, Herzog *et al.*, 1989). Thus, an important part of the evaluation of the potential impact on ground-water quality is an evaluation of a reasonable value (or more typically, a range of values) for the bulk hydraulic conductivity of any natural attenuation layer.

Hydraulic protection involves the use of natural ground-water levels (usually in the potential receptor aquifer) to induce a small flow into the landfill from the aquifer. Clearly, where there is ground-water flow into the landfill, there will not be an outward flow of leachate from the landfill to the aquifer. Also, the inward flow tends to reduce the outward movement of chemicals in the leachate due to the process of molecular diffusion. This concept of hydraulic protection (sometimes called a hydraulic trap) has gained popularity since the approval of the Halton Waste Management Facility (see Rowe *et al.*, 1993); however, as discussed by Rowe *et al.* (1994b) it is far simpler in concept than in implementation. In particular, it is important to consider not only the existing ground-water levels, but also the landfill base elevations, hydraulic conductivity of the aquitard and/or engineered system between the aquifer and the base of the waste, and the transmissivity of the aquifer to assess the effect of landfill construction and operation on water levels in the aquifer and the consequential potential impact on the effectiveness of the hydraulic trap.

In North America, the last decade has seen a major movement from largely uncontrolled disposal of waste in town dumps to the controlled disposal of waste in engineered landfills. The level of engineering can vary substantially, depending on the natural environment, the size of the landfill, the nature of the waste, and local regulations. As a minimum, most modern landfills have some form of engineered final cover over the waste that serves to control the infiltration of water into the waste and the consequent generation of leachate, as well as some form of engineered system for the collection of leachate. Some landfills have an engineered compacted clay liner to control the rate of migration of contaminants, others involve natural hydraulic protection combined with a backup compacted clay liner as an engineered contingency system (Rowe *et al.*, 1993; King *et al.*, 1993). In the United States composite liners are commonly used. These consist of a layer of plastic (typically 1-mm to 2-mm thick high-density polyethylene, HDPE) known as a geomembrane, overlying a compacted clay liner (*e.g.*, see Koerner, 1990; Rowe *et al.*, 1994b). A number of engineered systems will be discussed in the latter section of this paper.

Predictability of Ground-water Movement

It is important that the hydrogeology of a proposed landfill site be sufficiently well understood that it will allow reliable monitoring of the site. In addition, there needs to be some viable contingency measure that can be implemented in the event that some unexpected contamination of ground water does occur. This requirement for reasonable predictability is more restrictive with respect to what constitutes a suitable natural system than the requirement for protection of ground-water quality, since natural protection can be readily supplemented by additional engineering if needed. However, it is generally much more difficult to improve the predictability of a site using engineering methods.

Potential for Disruption of Ground-water Users

In this context, disruption of ground-water users includes both existing and potential users, particularly when there is no viable alternative water source. It may also involve potential disruption of

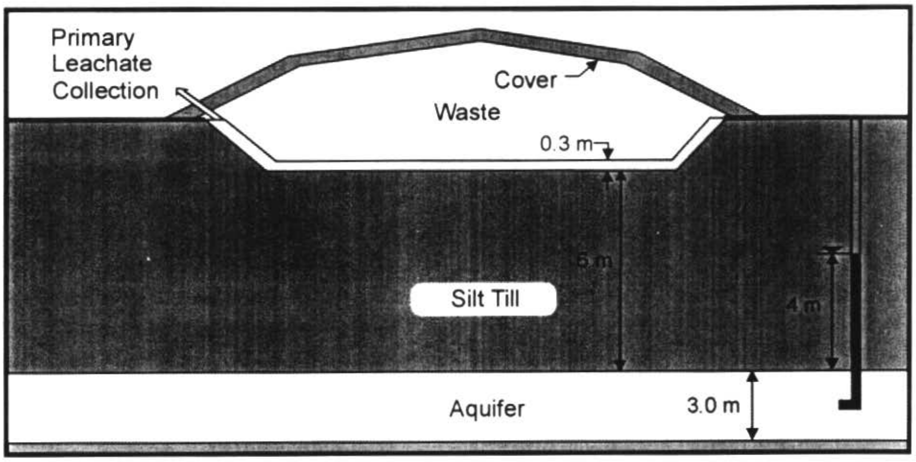

Figure 1 *Landfill design with leachate collection system only.*

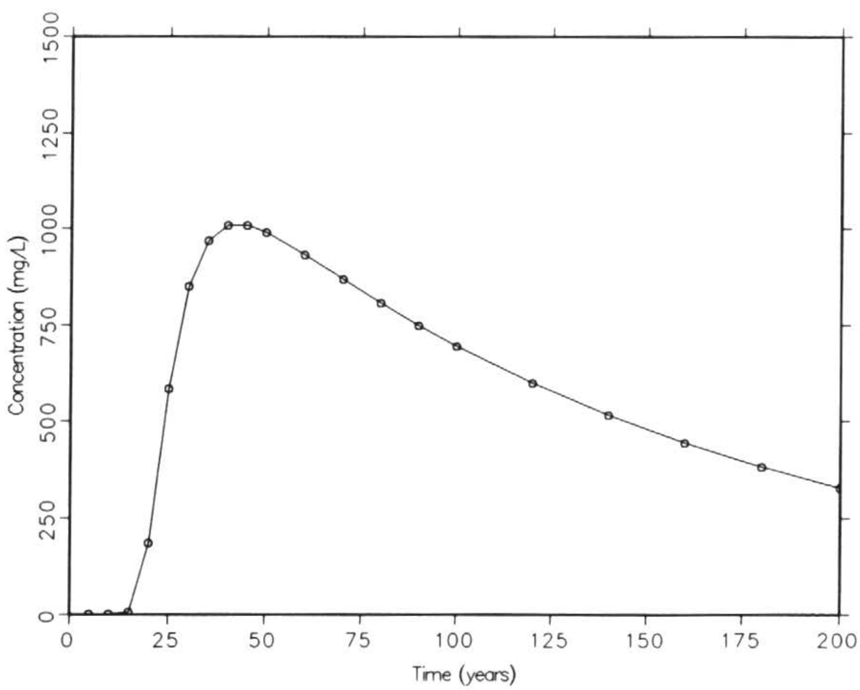

Figure 2 *Chloride concentration in aquifer for landfill design shown in Figure 1.*

stream baseflow due to ground-water drawdown.

The potential for disruption may be short term, either due to conventional construction drawdown or to the depressurization of an aquifer that can occur during construction of a hydraulic trap landfill. However, the potential disruption may also be long term, due to the cutoff of ground-water recharge over the area of the landfill, causing a drop in water levels, and/or due to a drop in water levels due to the operation of a hydraulic trap as discussed earlier.

A less obvious potential for disruption to ground-water users is a degradation in ground-water quality resulting from a change in water levels that induces mixing of unpotable water (*e.g.*, saline water in fractured bedrock) with what was originally overlying fresh water. This situation creates two potential problems. First, degradation of water quality is undesirable, irrespective of whether it results directly from leachate escaping from the landfill, or indirectly due to mixing of saline or brackish ground water with overlying fresh water. Second, this would complicate monitoring since chloride is one of the most common critical contaminants used to identify whether there has been an escape of leachate from a landfill. In this situation, it would be more difficult to identify whether an increase in chloride concentration was due to upwelling of underlying ground water or due to leachate escaping from the landfill. This problem is discussed further by Birks and Eyles, in press.

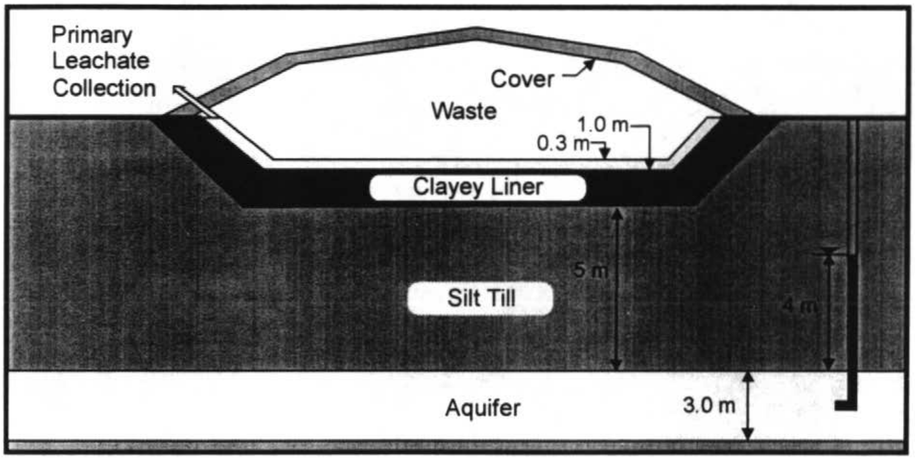

Figure 3 *Landfill with compacted clay liner.*

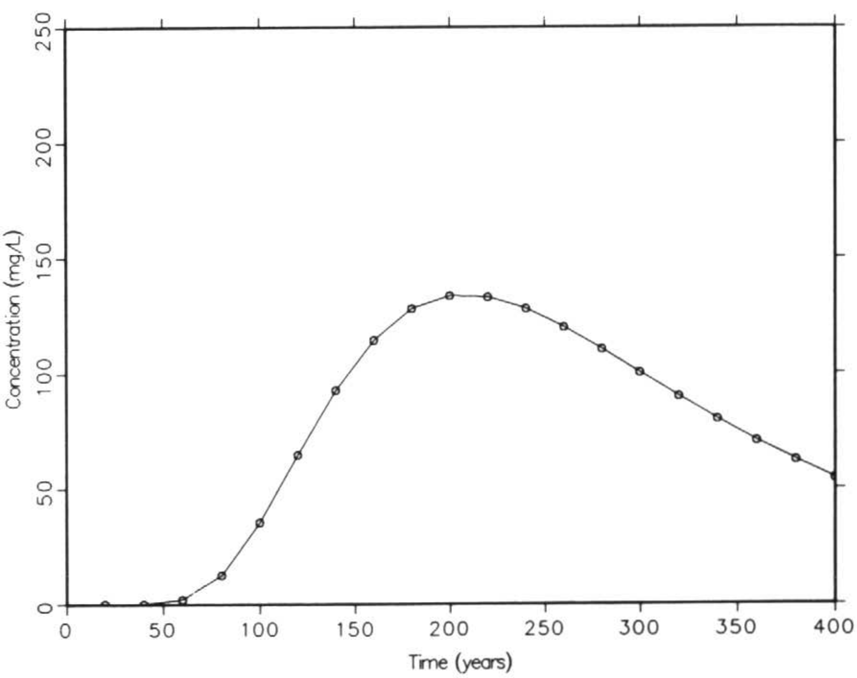

Figure 4 *Chloride concentration in aquifer for clay liner.*

Modelling

Observational techniques are used to establish existing site conditions. However, prediction of potential impacts often involves modelling which considers the interaction between the hydrogeology and the proposed engineering. The landfill design usually involves an interactive process wherein an initial design proposal is evaluated for its potential impact then revised, as necessary, to mitigate predicted impacts. For example, in the design of a hydraulic-trap landfill, the engineer can control the base elevations of the landfill, and the deeper these are below water levels in the underlying receptor aquifer, the greater will be the flow into the landfill (all other things being equal), and hence the better the hydraulic trap. However, there is a tradeoff between the benefits gained due to an increased ground-water gradient into the landfill and the disadvantages of decreasing the thickness of the attenuation layer between the landfill base and the receptor aquifer. Furthermore, there is an increased potential for disruption to ground-water users due to the volume of ground water collected, with the consequent changes in local ground-water levels. Alternatively, the engineer may examine different levels of engineering (*e.g.*, compacted clay *versus* composite liners, single *versus* double liners, *etc.*) when seeking to mitigate potential impacts.

Modelling will usually take the form of flow modelling and/or contaminant transport modelling. A detailed discussion on its application to engineered landfills is given by Rowe *et al.* (1994b). Flow modelling may range from hand calculations and simple analytical solutions (*e.g.*, Rowe and Nadarajah, 1994) to two-dimensional cross-sectioned models (*e.g.*, Frind and Matanga, 1985) and two-dimensional area models (*e.g.*, Franz and Guiger, 1989). Three-dimensional modelling (*e.g.*, Huyakorn *et al.*, 1986) is rarely used since the data base is often not sufficiently detailed to justify the high cost of performing three-dimensional analysis relative to the improvement in understanding that can be obtained. However, there are exceptions to this observation, and in some cases three-dimensional modelling can give valuable insights (*e.g.*, Molson and Frind, 1991; 1993; Livingstone *et al.*, in press).

Contaminant transport models vary substantially in sophistication and ease of use. A review of a number of commonly used models is given by Pandit *et al.* (1993) and Franz (1993), while Panigrahi *et al.* (1993) described the input requirements for many of these models. Franz and Rowe (1993) discuss the application of several models for a particular landfill design situation.

The following sections illustrate how simple models can be used to quickly evaluate the potential impact of different landfill designs on ground-water quality for a hypothetical case. The migration of contaminants from the landfill into the aquifer was modelled using a finite-layer analysis model (Rowe and Booker, 1985, 1991, 1994), as implemented in the computer program POLLUTEv6 (Rowe *et al.*, 1994a).

Since this impact is a consequence of the interaction between a particular hydrogeology and landfill design, the numerical results presented in this analysis should not be generalized beyond the level discussed in this paper.

EXAMPLE PROBLEM

In this analysis, the local hydrogeology is assumed to consist of a silty clay till overlying a gravel and sand aquifer (Fig. 1). The till is assumed to have a hydraulic conductivity of 1×10^{-8} $m \cdot s^{-1}$, a porosity of 0.4, and a diffusion coefficient of 0.02 $m^2 \cdot a^{-1}$. Beneath the till is a confined aquifer consisting of gravel and sand. This aquifer is assumed to be 3 m thick, and have a porosity of 0.35. At the up-gradient edge of the landfill, the horizontal flow in the aquifer per unit width is assumed to be 30 $(m^3 \cdot a^{-1}) \cdot m^{-1}$ (*i.e.*, a Darcy velocity of 10 $m \cdot a^{-1}$). This flow will be increased at the down-gradient edge of the landfill by the downward Darcy flux originating from the landfill. The potentiometric head in the landfill is assumed to be 4 m above the top of the aquifer. The infiltration through the silty landfill cover is assumed to be 0.15 $m \cdot a^{-1}$.

To quantify the impact associated with the interaction between the hydrogeology and the landfill design, the migration of chloride (a common component in municipal solid waste) was considered. The initial concentration after closure of the landfill was assumed to be 1500 $mg \cdot L^{-1}$, and the mass of the chloride was assumed to represent 0.2% of the waste. In this analysis, the waste was assumed to have an average thickness of 20 m and an apparent waste density of 600 $kg \cdot m^{-3}$. The landfill was assumed to be 1000 m long in the direction of ground-water flow. In assessing the impact of the landfill, the mass of contaminant was modelled as described by Rowe (1991a).

Some Landfill Design Considerations

The initial landfill design consists a 0.3 m-thick granular leachate collection system placed directly on top of the till (Fig. 1). In this and subsequent landfill designs, it is assumed that the till is excavated such that the base of the leachate collection system is 6 m above the top of the aquifer. This excavation allows for the placement of a 20-m thick waste pile.

Leachate is formed when rain water and runoff percolate through solid waste, leaching out soluble salts and biodegraded organic products. A leachate collection system is typically a granular layer with embedded pipes, used to collect and remove the leachate at the bottom of a landfill. The primary func-

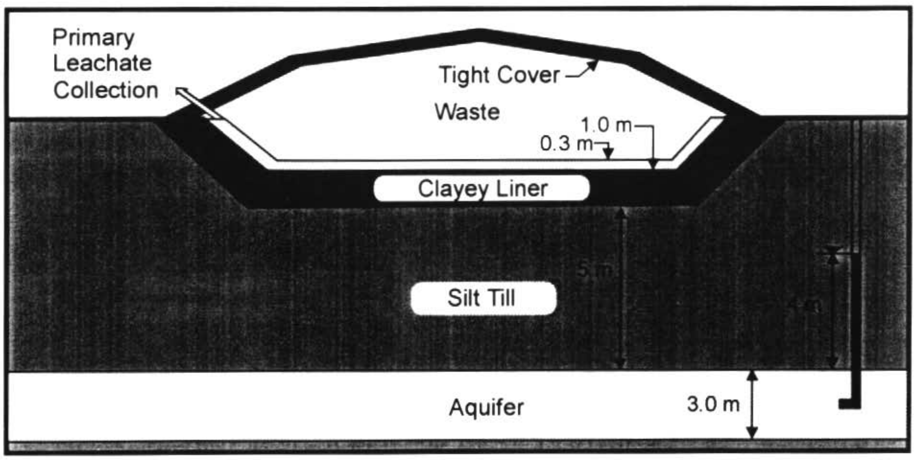

Figure 5 *Landfill with tight cover.*

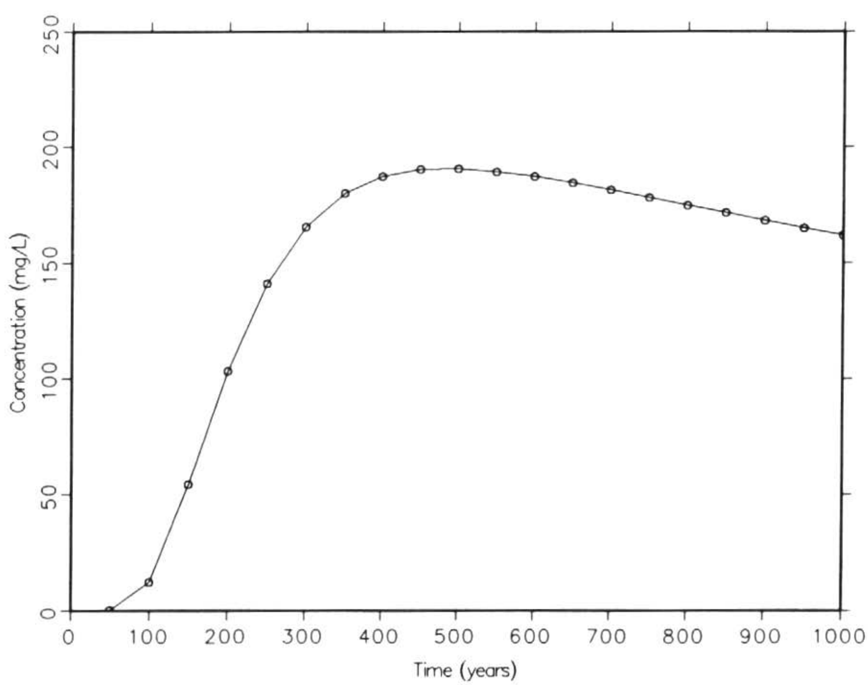

Figure 6 *Chloride concentration in aquifer for tight cover.*

tions of a leachate collection system are to reduce the volume of leachate in the landfill and, in particular, the pressure exerted by leachate at the base of the landfill. Removal of leachate also reduces the amount of contaminant available for transport into the hydrogeological system. By reducing the volume of leachate at the base of a landfill, the height of the leachate mound will be reduced, resulting in a lower hydraulic gradient beneath the landfill and, consequently, a lower Darcy velocity out of the landfill into the substrate. In this design, it is assumed that the leachate collection system is able to maintain the leachate mound at an average height of 0.3 m above the base of the landfill. The Darcy velocity beneath the landfill would then be 0.12 $m{\cdot}a^{-1}$, which would leave 0.03 $m{\cdot}a^{-1}$ (*i.e.*, about 25%) to be collected and removed by the leachate collection system.

Due to the downward Darcy velocity and diffusion, contaminants will migrate from the landfill through the till to the aquifer. As time passes, more and more contaminants will migrate to the aquifer at higher and higher concentrations. In this manner the mass of contaminants in the landfill is continuously depleted as contaminants are either removed by the leachate collection system or migrate downward. Because the mass of the contaminants is finite, the mass of contaminants transported into the aquifer will decline. Thus, there will be an initial increase in concentration in the aquifer, followed by a decline in concentration with time, creating a peak concentration in the aquifer (Birks and Eyles, in press).

Figure 2 shows the concentration of chloride in the aquifer that results from this landfill design and hydrogeology. The concentration in the aquifer reaches a peak value of about 1000 $mg{\cdot}L^{-1}$ at 45 years, and then declines. In Ontario, the Ministry of Environment and Energy Reasonable Use Policy (1994) limits the increase in the concentration of chloride in an aquifer to a maximum of 125 $mg{\cdot}L^{-1}$, assuming that there is negligible background concentration. According to this policy, the landfill design would not be acceptable.

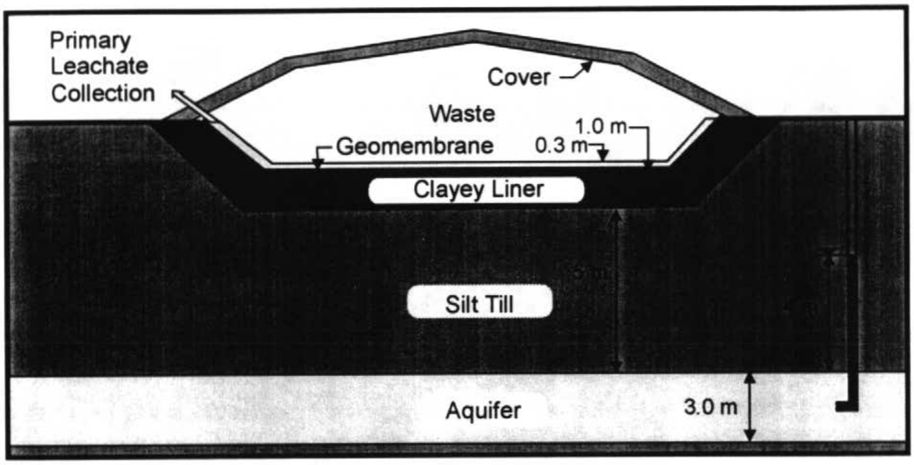

Figure 7 *Landfill design with composite liner.*

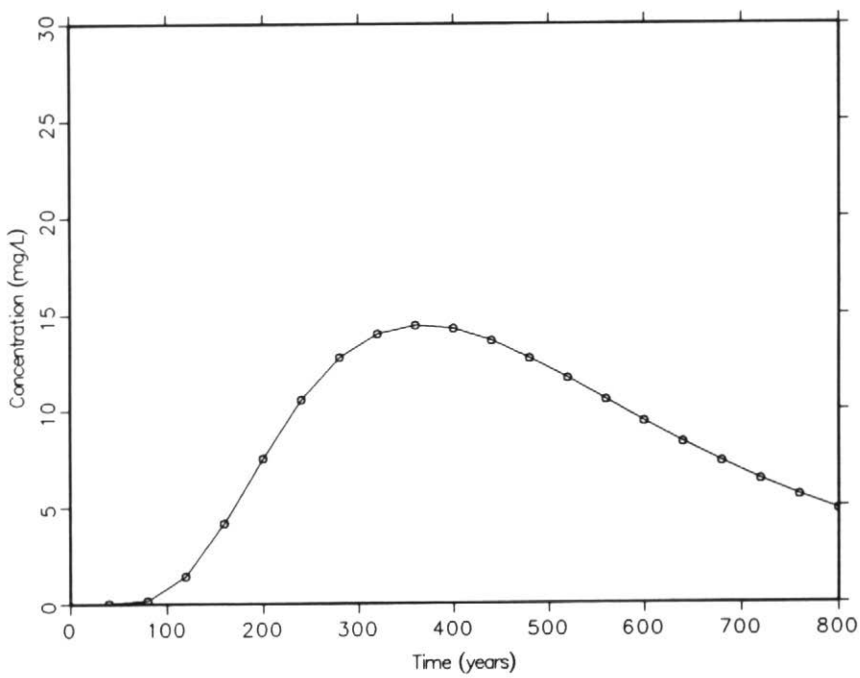

Figure 8 *Chloride concentration in aquifer for composite liner.*

Add a Clay Liner?

An alternative landfill design, that may result in a lower peak chloride concentration in the aquifer, would include a compacted clay liner beneath the primary leachate collection system (Fig. 3). This compacted clay liner would have a much lower hydraulic conductivity than the till, thus reducing the Darcy velocity beneath the landfill. In this analysis, the compacted clay liner is assumed to be 1 m thick, have a hydraulic conductivity of $2{\times}10^{-10}$ $m{\cdot}s^{-1}$, a porosity of 0.35, and a diffusion coefficient of 0.02 $m^2{\cdot}a^{-1}$ (Fig. 3). The resulting Darcy velocity beneath the landfill is now 0.013 $m{\cdot}a^{-1}$, instead of the previous 0.12 $m{\cdot}a^{-1}$. This lower Darcy velocity allows for the leachate collection system to function much more efficiently and collect about 91% of the leachate generated.

The concentration of chloride in the aquifer that would result from this landfill design is shown in Figure 4. This concentration reaches a maximum value of 133 $mg{\cdot}L^{-1}$ at 200 years, which is still above the maximum 125 $mg{\cdot}L^{-1}$ allowed by the Ontario Ministry of the Environment and Energy (OMOEE).

Add a Tight Cover?

A possible design change that might be considered is to add a low-permeability (tight) cover over the landfill (Fig. 5). By adding a tight cover, the amount of percolation through the waste is limited, resulting in less leachate being produced each year. This tight cover is assumed to limit the infiltration into the landfill to 0.008 $m \cdot a^{-1}$, which will control the maximum Darcy velocity that can occur beneath the landfill. Thus the Darcy velocity beneath the landfill is 0.008 $m \cdot a^{-1}$, and the amount of leachate that is collected by the leachate collection system is negligible.

Figure 6 shows the concentration of chloride in the aquifer for a landfill design that incorporates a tight cover. Notice that a significant amount of contaminant reaches the aquifer. These contaminants are primarily transported by the process of molecular diffusion, since the Darcy velocity is low due to the tight cover. In this design, the maximum chloride concentration was 190 $mg \cdot L^{-1}$ at 500 years, which is even higher than that for the design with a permeable cover. By adding a tight cover, the peak concentration in the aquifer was delayed by 300 years, since diffusion tends to be a slower process than advection. The magnitude of the peak concentration increased since very little contaminant was removed by the leachate collection system.

Add a Geomembrane Liner?

Another possible design alternative that may reduce the amount of contaminants reaching the aquifer is to add a geomembrane on top of the compacted clay liner. This type of barrier is called a composite liner (Fig 7). The geomembrane in this design is assumed to be 1.5-mm thick high-density polyethylene (HDPE), and have a diffusion coefficient of 3×10^{-5} $m^2 \cdot a^{-1}$. During the manufacture and installation of a geomembrane, small holes or defects may be introduced into the geomembrane.

The geomembrane is assumed to have small defects of 0.1 cm^2 area with a frequency of one every acre (2.5/ha). The effective hydraulic conductivity of the geomembrane is then 1.1×10^{-15} $m \cdot s^{-1}$, which is based upon the likely leakage through a well-constructed composite liner using information from Giroud *et al.* (1992). The Darcy velocity through the composite liner and silt till would be 5.3×10^{-5} $m \cdot a^{-1}$. The volume of leachate that would be collected by the leachate collection system, assuming a permeable cover, would be 0.1499 $m \cdot a^{-1}$ (*i.e.*, essentially 100%).

The concentration of chloride in the aquifer that would result from this design, incorporating a composite liner, is shown in Figure 8. A maximum chloride concentration of 14 $mg \cdot L^{-1}$ occurs at 360 years. This maximum is well below the maximum 125 $mg \cdot L^{-1}$ specified by OMOEE.

What If the Collection System Clogs?

The time period during which an engineered leachate collection system is fully functional is defined as its service life. The service life is highly dependent on the design of the system (Rowe, 1991a, b). For example, leachate collection systems may eventually clog due to chemical and biological activity.

While the leachate collection system is functioning, the leachate mound at the base of the landfill is likely to be relatively small, in this design it is assumed to be an average of 0.3 m. If the leachate collection system fails and be-

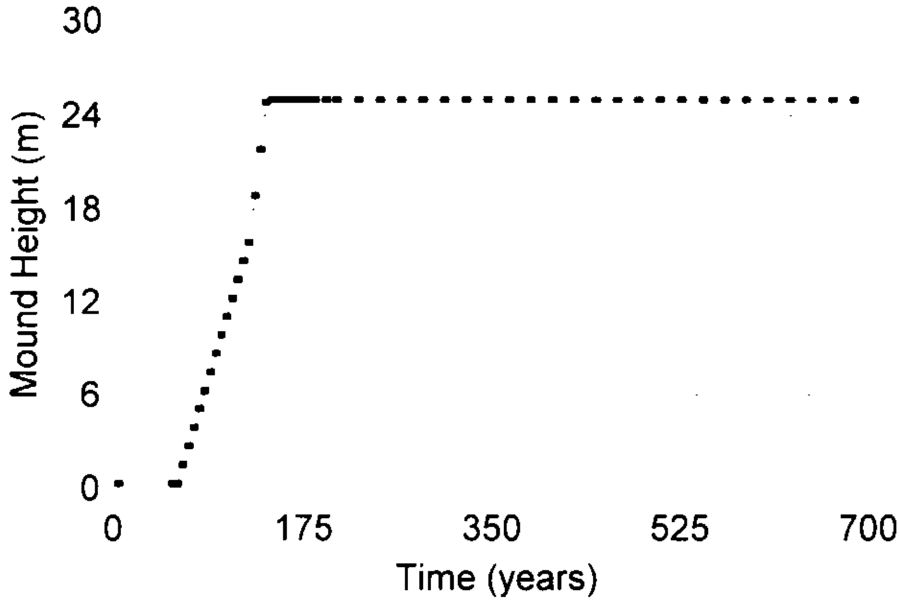

Figure 9 *Leachate mound when leachate collection system fails.*

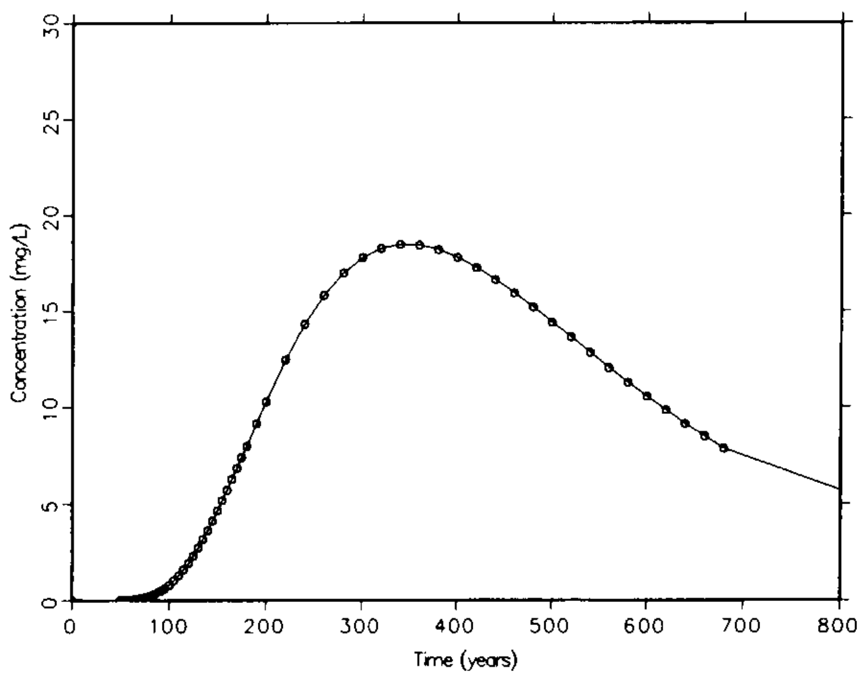

Figure 10 *Chloride concentration in aquifer for failed leachate collection system.*

comes clogged, the leachate mound will increase in height at a rate controlled, *inter alia*, by the infiltration through the cover and the downward Darcy velocity through the liner. The maximum height of the leachate mound is also controlled by the thickness of the waste, in this analysis assumed to average 25 m. If the leachate mound reaches this maximum height, any excess leachate generated will escape from the landfill *via* toe drains and seeps through the cover.

The service life of the leachate collection system in this analysis is assumed to be 50 years after closure. After this time, the leachate collection system begins to experience significant decreases in performance due to clogging, until at 75 years it is no longer controlling the height of the leachate mound in the landfill (Fig. 9).

In Figure 10, the resulting chloride concentration in the aquifer is shown, assuming the leachate collection system fails. The maximum chloride concentration in the aquifer is 18 mg·L^{-1} at 340 years, which is only slightly more than it was when the leachate collection system did not fail. Thus, it would appear that the failure of the leachate collection system is not a major concern, assuming that the geomembrane has an infinite service life.

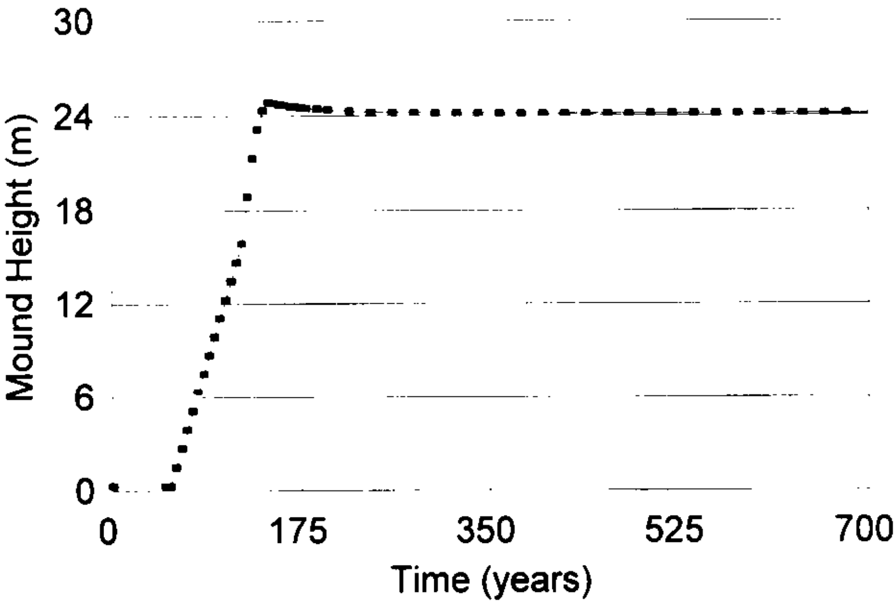

Figure 11 *Leachate mound when both leachate collection system and geomembrane fail.*

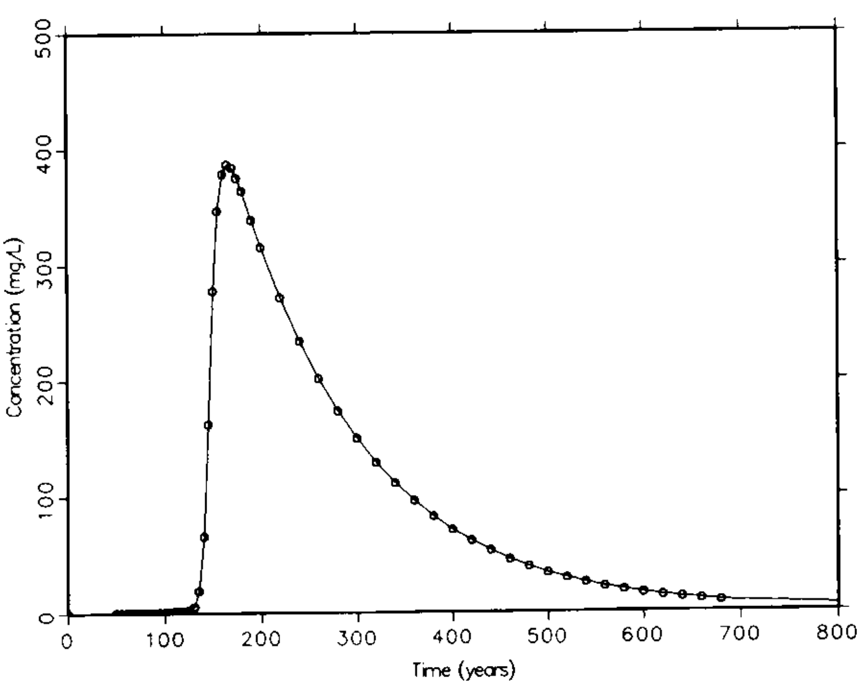

Figure 12 *Chloride concentration in aquifer for failed leachate collection system and geomembrane.*

What If Geomembrane Degrades?

Geomembranes have a limited service life, due to degradation caused by chemical attack and other processes. This results in an increase in the effective hydraulic conductivity of the geomembrane. In this analysis, the geomembrane is assumed to have a service life of 125 years, after which it will begin to significantly degrade, until, at 150 years, it is no longer having an impact upon the Darcy velocity beneath the landfill. As the geomembrane degrades, the Darcy velocity will start to increase until it reaches a maximum value which is assumed to be that of the compacted clay beneath the geomembrane. These changes in Darcy velocity will have an effect upon the height of the leachate mound. Initially, the height of the leachate mound will be 0.3 m while both the leachate collection system and geomembrane are functioning. After the leachate collection system fails, this mound will increase to its maximum height, where it will stabilize until the geomembrane fails. When the geomembrane fails, the leachate mound may decrease in height due to the increased Darcy velocity through the landfill liner. Eventually, the leachate mound will stabilize at a new height that is controlled by the Darcy velocity through the liner and the infiltration through the cover (Fig. 11).

The calculated chloride concentration in the aquifer is shown in Figure 12 for this design, assuming finite service lives of the leachate collection system and geomembrane. Based upon these assumptions, the maximum chloride concentration in the aquifer is 387 mg·L^{-1} at 165 years, which would not be acceptable according to the OMOEE. At this stage, it would be necessary to further refine the design of the landfill to achieve a contaminant impact that is acceptable according to the OMOEE policy. These refinements may include 1) addition of a secondary leachate collection system and liner, 2) use of a lower permeability compacted clay liner, 3) changes in the base elevation of the landfill, and/or 4) control of the leachate mound after failure of the leachate collection system.

DISCUSSION

Irrespective of how much engineering is proposed, it is important to have an adequate understanding of site geology and hydrogeology to allow confident monitoring of the site and the development of contingency measures that could be used to mitigate any unexpected escape of leachate from the facility. The engineered design does not reduce the need for an adequate hydrogeological investigation. Most modern landfills will require some form of engineering, and the interaction between this engineering and the natural system also needs to be considered by means of flow and/or contaminant transport modelling. Consideration should also be given to the service life of the components of this system and the implications that this may have on potential impacts on ground-water quality.

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Accepted, as revised, January 1996.