

# Environmental issues in geotechnical engineering

## Problèmes d'environnement en génie géotechnique

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### ABSTRACT

Although the exact date of when geotechnical engineers began to address environmental issues is uncertain, such issues became a primary concern in the field of Geotechnical Engineering around 1980. Thus, geotechnical engineers have been dealing with environmental issues on a formal basis for at least a quarter of a century. During the interim period, the significance of environmental issues in Geotechnical Engineering has grown to the extent such that a new sub-discipline in Geotechnical Engineering, commonly referred to as Environmental Geotechnics, has been established. As a consequence of these events, an assessment of some environmental issues that are currently of interest in Geotechnical Engineering, as well as some issues that are likely to be important in the future, is appropriate at this time. Accordingly, after a brief historical perspective, seven current and/or future environmental issues in Geotechnical Engineering are described, including the (1) long-term performance of waste containment systems, (2) acceptance of alternative barriers and barrier materials, (3) need for innovative barriers and innovative barrier materials, (4) emergence of new waste forms, (5) increasing significance of biological processes, (6) role of modeling, and (7) importance of professional identity. The significance of each issue is illustrated through one or more examples. The overall outcome of the assessment is that environmental issues will continue to represent a major sub-category within Geotechnical Engineering in the foreseeable future.

### RÉSUMÉ

Bien que la date exacte à laquelle les ingénieurs géotechniques ont commencé à s'occuper des problèmes d'environnement soit incertaine, ces problèmes sont devenus des soucis prioritaires vers 1980. Les ingénieurs géotechniques s'occupent officiellement des problèmes d'environnement depuis un quart de siècle. Pendant la période intérimaire, l'importance des problèmes d'environnement dans le génie géotechnique est devenue telle qu'elle a engendré une nouvelle sous-discipline du génie géotechnique que l'on nomme généralement géotechnique de l'environnement. Suite à ces événements, une évaluation des problèmes d'environnement en génie géotechnique, ainsi que des problèmes qui prendront vraisemblablement de l'importance à l'avenir s'impose à présent. En conséquence, après une brève perspective historique, on décrit sept problèmes présents (et) ou futurs en génie de l'environnement: (1) performance à long terme du système d'endiguage des déchets (2) acceptation de barrières et de matériaux d'endiguage différents (3) nécessité de créer des barrières et des matériaux d'endiguage novateurs (4) émergence de nouveaux types de déchets (5) importance accrue des processus biologiques (6) rôle des simulations et (7) importance de l'identité professionnelle. On a illustré l'importance de chaque problème à l'aide d'un ou plusieurs exemples. La conclusion finale de cette évaluation est que les problèmes d'environnement continueront à représenter une sous-catégorie majeure du génie géotechnique dans un avenir prévisible.

### 1 INTRODUCTION

Environmental issues have been a significant component of Geotechnical Engineering since about 1980, although geotechnical engineers likely have been dealing with environmental issues on a less formal, more piecemeal basis over a substantially longer time frame. The primary environmental issues facing geotechnical engineers around 1980 pertained to (1) the design and construction of new waste containment facilities, such as engineered landfills used for disposal of municipal solid waste (MSW) and hazardous waste, and (2) the assessment and remediation of sites contaminated by past industrial practices and posing a threat to the public health and the environment. Some of the main issues at that time dealt with our ability to accurately measure the hydraulic conductivity ( $k$ ) of all types of geomaterials, including natural aquitards, compacted clay liners (CCLs), vertical barriers (e.g., cutoff walls), and covers used in waste containment systems, the effects of chemical solutions (e.g., leachates) on the  $k$  of soils (referred to as "compatibility"), the role of contaminant transport through CCLs, and the problems associated with poor or marginal construction quality control for engineered containment systems. Concerns over the integrity of CCLs arose due to problems related to such factors as incompatibility between CCLs and chemical solutions resulting in significant increases in  $k$ , and protection of CCLs from envi-

ronmental distress (e.g., desiccation cracking, freezing and thawing) both during and after construction. Such concerns led to the advent of the use of composite liners, whereby relatively thin (0.76 – 2.3 mm) polymer materials, known as flexible membrane liners (FMLs) then and as geomembrane liners (GMLs) today, were placed on top of and in intimate contact with CCLs as a single liner. The primary purpose of composite liners was to provide some redundancy whereby the relatively thin, but essentially "impermeable" GML was complemented by the more permeable, but thicker CCL.

Towards the end of the 1990s, geotechnical engineers continued to focus on scale effects related to the measurement of the  $k$  of CCLs in the field relative to the laboratory, as well as other issues associated with waste containment systems, such as the compressibility of MSW and our ability to predict the settlement of MSW in landfills. The advent of and interest in the use of a wide variety of geosynthetic materials other than GMLs, such as geotextiles, geonets, and geocomposites, began to increase around 1990, and the geosynthetic clay liner (GCL), which consists of a thin layer of sodium bentonite sandwiched between two geotextiles and held together by needle punching, stitching, and/or gluing, was introduced into the market. Although the roles of contaminant transport, in general, and diffusion, in particular, through liner materials previously had been recognized, these issues came to the forefront around 1990.

Geotechnical engineers began to understand that their ability to address issues related to contaminant transport and chemical compatibility relied to a large extent on their understanding of concepts in the fields of chemistry and groundwater hydrology, and the need to broaden the formal educational backgrounds of geotechnical engineers to include some of these and other concepts was becoming recognized.

In the 1990s, as the engineering aspects of liners and covers for waste containment facilities matured, efforts were undertaken to improve (i) our understanding of the operation of leachate collection and removal systems at landfills, (ii) our ability to predict waste settlement, and (iii) our knowledge of the properties of geosynthetics used in waste containment facilities. Issues related to the stability of wastes and waste containment systems under both static and dynamic loading conditions also came to the forefront during the 1990s.

From the late 1970s to early 1990s, geotechnical engineers also started to focus their efforts on addressing issues related to assessment and remediation or clean-up of contaminated sites. These efforts involved cooperation with other disciplines, such as Geology, Chemical Engineering, and Environmental Engineering. While professionals from these other disciplines may have possessed a greater knowledge of specific chemical and biological processes inherent in many of the treatment technologies being developed at that time, their experience dealt primarily with manufactured or "clean" treatment systems as opposed to soil or "dirty" systems. Thus, as Civil Engineers, geotechnical engineers were able to contribute their knowledge of civil engineering systems, in general, and soil behavior, in particular, to the development and assessment of treatment systems for remediation of contaminated sites.

In particular, geotechnical engineers' long record of experience in dealing with the behavior of clays as engineered materials provided the impetus for substantial research in the 1990s aimed at using electrical fields for the removal of contaminants from clays (e.g., via electro-osmosis). Geotechnical engineers also have readily adapted their experience with ground modification techniques used to address environmental issues related to remediation, such as the stabilization and solidification of contaminants in the subsurface (e.g., via shallow and deep soil mixing using stabilizers such as cement and lime). Also, geotechnical engineers' long history with the use of vertical cutoff walls for structural purposes provided experience that was readily adaptable to the use of vertical cutoff walls, such as soil-bentonite and cement-bentonite cutoff walls, for *in situ* containment of contaminated ground water. At the same time, with their increasingly more diverse formal education, geotechnical engineers began to play an increasingly greater role in the development of other *in situ* treatment systems for remediation, such as the use of surfactants or co-solvents for mobilizing and removing nonaqueous phase liquids (NAPLs) from the subsurface, and the use of zero-valent iron in passive reactive treatment walls (i.e., trenches) for dechlorination of chlorinated solvents, such as tetrachloroethylene (PCE) and trichloroethylene (TCE), in ground water contaminated with these compounds.

As is evident from this brief, historical perspective, geotechnical engineers have played a significant role in addressing environmental issues on a formal basis for at least a quarter of a century. During this interim period, the significance of environmental issues in Geotechnical Engineering has grown to the extent such that a new sub-discipline in Geotechnical Engineering, commonly referred to as Environmental Geotechnics, has been recognized. In recognition of the historical impact of environmental issues on Geotechnical Engineering, the objective of this paper is to illustrate some of the current and future environmental issues facing Geotechnical Engineering. In this regard, seven current and/or future environmental issues in Geotechnical Engineering are described, including the (1) long-term performance of waste containment systems, (2) acceptance of

alternative barriers and barrier materials, (3) need for innovative barriers and innovative barrier materials, (4) emergence of new waste forms, (5) increasing significance of biological processes, (6) role of modeling, and (7) importance of professional identity and the need for consistent professional terminology. The significance of each issue is illustrated through one or more examples.

## 2 LONG-TERM PERFORMANCE OF WASTE CONTAINMENT SYSTEMS

The exact date for the advent of the "modern" waste containment system is uncertain, but the use of liners in waste containment systems to protect groundwater quality has been practiced for some types of landfills in some parts of the United States from about the mid 1970s (Bonaparte et al., 2002). This time frame also corresponds to some extent with the promulgation in the United States of the Resource Conservation and Recovery Act (RCRA) in 1976 regulating the disposal of solid wastes (RCRA Subtitle D) and hazardous wastes (RCRA Subtitle C). Since then, the use of waste containment systems in landfills, surface impoundments, and waste piles has become increasingly more widespread, and the complexity and capabilities of these systems have improved progressively. In fact, disposal in landfills containing engineered waste containment systems remains the most widely used method for dealing with MSW and many other types of waste in the United States (Bonaparte et al., 2002).

A relatively recent review of the state of the practice of modern waste containment systems illustrates that such systems are to date a success story in terms of minimizing the potential for groundwater contamination (Benson and Edil, 2004). Although the liners of modern waste containment systems are known to leak, the leakage rates typically are sufficiently low such that mass discharges of contaminants are limited to rates that allow attenuation of the contaminants by natural environmental processes without significant detrimental impacts to human health and the environment. However, leachate generated by landfills in the United States is required to be collected for the active life of the landfill plus a 30-year post-closure period. Given the relatively young age of modern waste containment systems, and the periodic modification of the regulations governing the disposal of wastes, this 30-year period has yet to be reached for any landfill constructed in the United States under current U.S. Environmental Protection Agency (EPA) guidelines (Bonaparte et al., 2002). As a result, continuous assessment of modern waste containment facilities will be required to ensure that the long-term performance of such facilities continues to protect groundwater quality, the environment, and human health. Therefore, the long-term performance of modern waste containment facilities will continue to be an environmental issue of concern to geotechnical engineers for the foreseeable future.

In this regard, two examples of some recent data are provided to illustrate the potential importance of this issue, viz, (1) the relatively recent occurrence of volatile organic compounds (VOCs) in collection lysimeters beneath liners in a number of landfills in Wisconsin, USA, and (2) the hydraulic conductivity of geosynthetic clay liners (GCLs) subjected to extended permeation with inorganic salt solutions.

### 2.1 VOCs in Wisconsin landfills

The Wisconsin Department of Natural Resources (WDNR) is relatively unique in the United States in that the WDNR historically has required the installation of large (~ 10 m by 10 m) collection lysimeters (underdrains) directly beneath the liner of each lined landfill cell to monitor the quantity and quality of water discharged through the liner (see Fig. 1). As illustrated in

Fig. 2, recent analysis of the liquid collected in some of these lysimeters indicates that a wide variety of VOCs in various concentrations have appeared at different frequencies in 91 lysimeters. Concentrations greater than maximum contaminant levels (MCLs) have been found in 90 of the 1200 samples (~ 8 %) from cells with liners containing geomembranes, primarily for toluene, tetrahydrofuran, dichloromethane, benzene, and ethylbenzene (Benson and Edil, 2004).

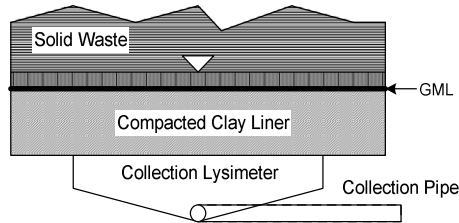


Figure 1. Schematic of typical collection lysimeter (underdrain) beneath a compacted clay liner for a solid waste disposal facility.

In general, temporal trends in the VOC concentrations have not been evident, but rather concentrations tend to fluctuate around some mean value (Benson and Edil, 2005). The temporal data in dichloromethane (DCM) concentrations in Fig. 3 exhibit more trend than most. Another interesting aspect of the data collected thus far is that the VOC concentrations in the lysimeters tend to be on average about 5 to 10 times less than the concentration in the leachate (Benson and Edil, 2005).

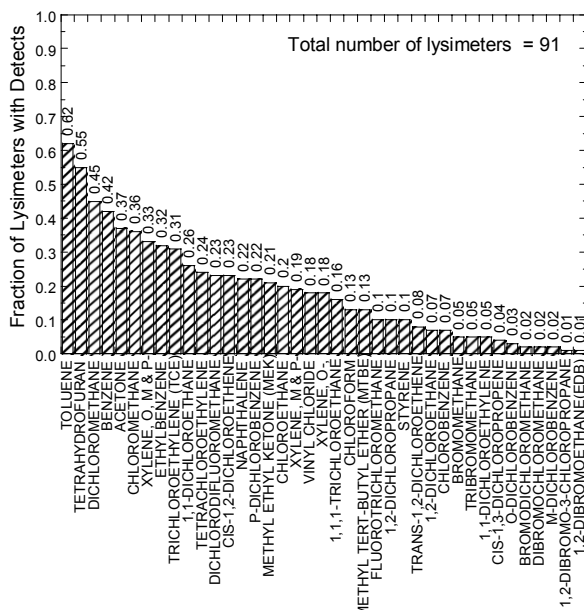


Figure 2. Fraction of collection lysimeters beneath landfills in Wisconsin (USA) with compacted clay or geomembrane-compacted clay liners containing volatile organic compounds (data from Benson and Edil, 2005).

Finally, the concentrations of dichloromethane (DCM) collected in lysimeters beneath composite lined cells are compared with those from clay lined cells in the box plots shown in Fig. 4. The center line in each box plot represents the median of the data, the outer edges of each box represent the interquartile range (i.e., 25<sup>th</sup> to 75<sup>th</sup> percentiles), and the outermost lines or "whiskers" represent the 5<sup>th</sup> and 95<sup>th</sup> percentiles (e.g., Albright et al., 2004). As shown in Fig. 4, the concentrations of DCM in collection lysimeters beneath composite lined cells do not tend to be any lower than those collected beneath cells lined only

with compacted clay. This similarity in DCM concentrations is not necessarily surprising, given that geomembranes typically provide little resistance to diffusion of VOCs (e.g., Edil, 2003).

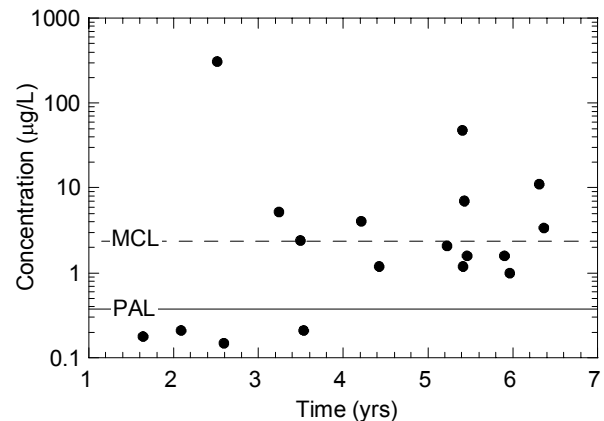


Figure 3. Temporal concentrations of dichloromethane (DCM) in a collection lysimeter (underdrain) beneath a composite lined waste disposal cell in a landfill in Wisconsin; PAL = protective action limit (data from Benson and Edil, 2005).

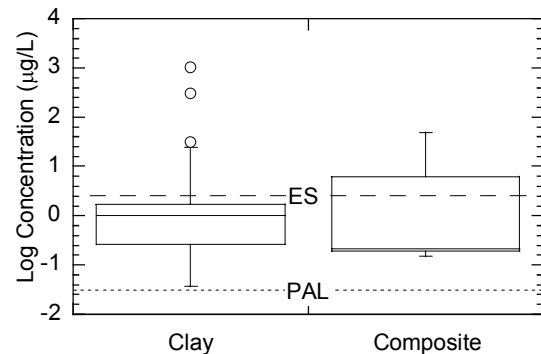


Figure 4. Box plot comparisons of dichloromethane (DCM) concentrations in collection lysimeters beneath composite lined and clay lined cells in landfills in Wisconsin; ES = enforcement standard; PAL = protective action limit (data from Benson and Edil, 2005).

The existence of VOCs in these lysimeters was not necessarily anticipated, primarily because the liners in Wisconsin have traditionally been thicker than those required by federal regulations (i.e., RCRA Subtitle D for MSW disposal facilities). For example, prior to 1996, the WDNR required landfill cells to be lined with a minimum of 1.524 m of compacted clay, which was 67 % greater than the minimum thickness of 0.914 m of compacted clay required by RCRA Subtitle D at that time (see Fig. 5). Subsequent to 1996, the WDNR has required a composite liner consisting of a geomembrane liner (GML) overlying and in intimate contact with 1.219 m of compacted clay. Although this requirement for a composite liner is consistent with the current RCRA Subtitle D requirements (Federal Register 1991), the minimum thickness for the compacted clay portion of the composite liners in Wisconsin is still 100 % thicker (1.219 vs. 0.610 m) than that required by Subtitle D. Nonetheless, the very low concentrations observed to date suggest that the liners in modern engineered landfills in Wisconsin have functioned well, at least for the relatively short time frame of performance (~ 10 to 20 yrs), and the effects of natural attenuation processes such as dilution, adsorption, and degradation likely will render concentrations of these VOCs below detectable levels by the time the contaminants reach a regulatory compliance point, such as a perimeter monitoring well (Benson and Edil, 2004). However, the performance record is still relatively short such that continued

monitoring of these and other landfills will be required over the longer term before any final conclusions can be drawn regarding the performance of modern waste containment systems.

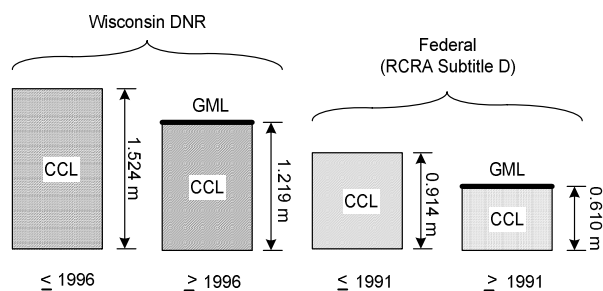


Figure 5. Comparison of requirements for Wisconsin Department of Natural Resources (WDNR) landfill liners with those based on U. S. federal regulations for solid waste disposal (CCL = compacted clay liner; GML = geomembrane liner).

## 2.2 Cation exchange and long-term hydraulic conductivity of GCLs

Manufactured geosynthetic clay liners (GCLs) comprised of thin sheets (~ 10 mm) of sodium bentonite sandwiched between two geotextiles and held together by needle-punching, stitching, and/or gluing, have become increasingly preferred for use in liners and covers for waste containment systems, primarily due to ease of construction and relatively low cost. However, the results of several studies have shown that calcium-for-sodium cation exchange can result in reduced swelling capability of the bentonite in the GCLs upon hydration and ultimately to desiccation of the bentonite and poor hydraulic performance of the GCLs (*Aboveground Tank Update*, 1992; James et al., 1997; Shackelford et al., 2000; Egloffstein, 2001; Jo et al., 2001, 2004; Benson, 2002). The calcium is derived from surrounding soils, and migrates into the GCL usually under unsaturated conditions presumably in response to both hydraulic (suction) and chemical (diffusive) gradients.

For example, a GCL used to line a containment facility for an oil storage tank was found desiccated (*Aboveground Tank Update*, 1992). Extraction of the exchange complex of the bentonite in the GCL revealed that the sodium cations that originally dominated the exchange complex had been replaced by calcium cations presumably via flow or diffusion from the underlying foundation soil under unsaturated conditions. In other cases, exhumation of GCLs placed in cover systems has resulted in the observation that the GCLs have become extremely desiccated (e.g., see Fig. 6) due to cation exchange resulting from leaching of calcium rich soils placed over the GCLs for protection (James et al., 1997; Benson, 2002). The desiccation results because the swelling capacity of the bentonite is reduced significantly when the exchange complex is dominated by calcium cations, such that re-hydration of the bentonite does not result in sufficient swelling to close the cracks. Although post-construction forensic analyses revealed the cause of the desiccation and the resulting poor performance of the GCLs, no fundamental understanding of the mechanisms associated with the migration of calcium (or other multivalent cations) has been gained. Such an understanding may be important, for example, in terms of determining measures to prevent such desiccation from occurring *a priori* (i.e., during design).



Figure 6. Photograph of desiccated bentonite in a GCL exhumed from a cap in southwestern Wisconsin (from Benson, 2002).

The effect of invading salts (i.e., cations) also has important implications in terms of the long-term hydraulic performance of GCLs used as liners or components of liners in waste containment facilities. For example, consider the data shown in Fig. 7 based on hydraulic conductivity testing of GCLs permeated with chemical solutions containing different concentrations of calcium chloride ( $\text{CaCl}_2$ ). The non-prehydrated specimens were exposed to the  $\text{CaCl}_2$  solutions for 48 hrs prior to the start of permeation with the  $\text{CaCl}_2$  solutions, whereas the prehydrated specimens were permeated with deionized water for extended durations (> 1 yr) prior to permeation with the  $\text{CaCl}_2$  solutions. For all tests, the hydraulic gradient ( $i$ ) applied for permeation was approximately 200, and all of the tests were conducted until chemical equilibrium between the effluent and influent in terms of calcium ( $\text{Ca}^{2+}$ ), chloride ( $\text{Cl}^-$ ), and electrical conductivity (EC) had been achieved.

As shown in Fig. 7, the hydraulic conductivity of all GCL specimens increased by the time chemical equilibrium had been established, with greater increases in hydraulic conductivity and shorter test durations occurring with an increase in  $\text{CaCl}_2$  concentration. In the case of the tests performed using solutions with the lower, 5, 10, and 20 mM  $\text{CaCl}_2$  concentrations, the test durations required to achieve chemical equilibrium ranged from about 0.5 to 1.5 yrs for the non-prehydrated specimens and from about 1.3 to 2 yrs for the prehydrated specimens. The longer test durations for the tests performed using solutions with the lower  $\text{CaCl}_2$  concentrations can be attributed, in part, to the lower rate of  $\text{Ca}^{2+}$  mass loading such that the time required for equilibrium exchange with the sodium ( $\text{Na}^+$ ) initially in the exchange complex of the bentonite increases (Jo et al., 2005). The longer test durations for the prehydrated specimens relative to the non-prehydrated specimens can be attributed, in part, to the initially lower hydraulic conductivity obtained as a result of swelling of the bentonite due to permeation with deionized water (although this observation is not necessarily readily apparent for the tests using the relatively dilute 5, 10, and 20 mM  $\text{CaCl}_2$  solutions). As shown in Fig. 8, the quality of the bentonite in the GCL, where higher quality is reflected by greater montmorillonite content, higher cation exchange capacity, higher liquid limit and plasticity index, and greater swelling potential, also affects the test duration in the case of permeation with solutions containing relatively high  $\text{CaCl}_2$  concentrations, but has relatively little, if any, effect for tests performed using relatively dilute  $\text{CaCl}_2$  solutions.

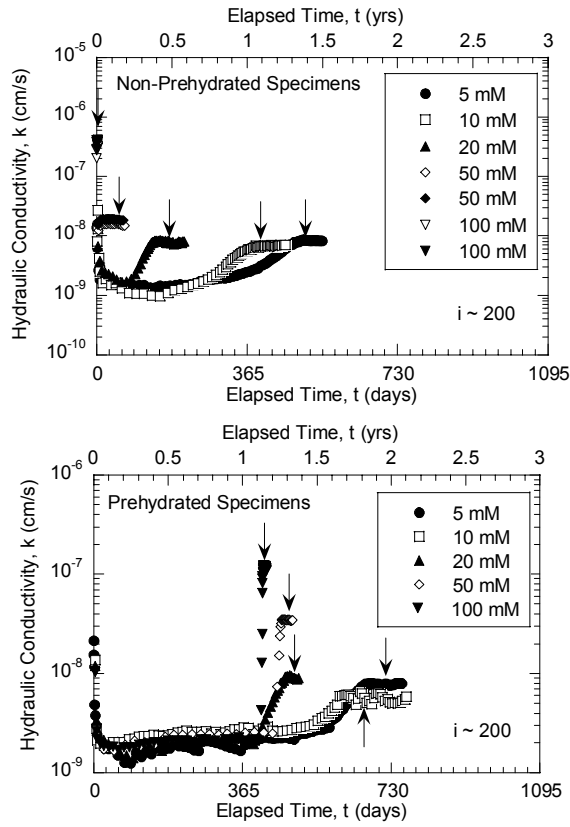


Figure 7. Temporal trends in hydraulic conductivity of non-prehydrated and prehydrated GCLs permeated with calcium chloride ( $\text{CaCl}_2$ ) solutions; time corresponding to establishment of chemical equilibrium designated by arrows (data from Lee, 2004).

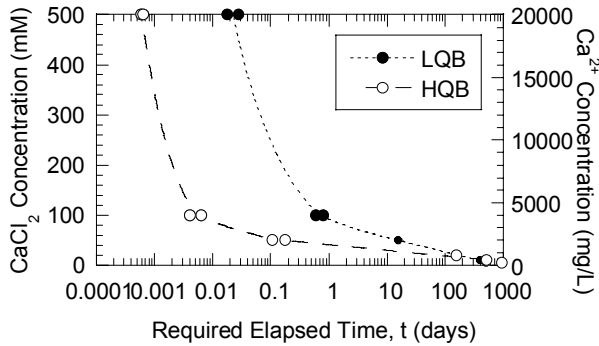


Figure 8. Test durations for hydraulic conductivity tests based on chemical equilibrium between effluent and influent for specimens of geosynthetic clay liners containing lower quality bentonite (LQB) or higher quality bentonite (HQB) as a function of the permeant liquid concentration (data from Lee and Shackelford, 2005).

The test durations shown in Figs. 7 and 8 are based on laboratory permeability testing using a relatively high hydraulic gradient ( $i \sim 200$ ) such that the times required to achieve chemical equilibrium in field situations, where much lower hydraulic gradients are expected, are likely to be significantly greater. On the basis that the time required to achieve chemical equilibrium and steady-state hydraulic conductivity is a function of the salt mass loading rate, primarily the salt cations (e.g.,  $\text{Ca}^{2+}$ ), the times required to achieve chemical equilibrium for a GCL in the field can be evaluated via the mass loading rate, which for a constant source solution containing a single salt is given as follows:

$$\left(\frac{dV}{dt}\right)_{c_0} = k(t) \cdot i \cdot A \cdot c_0 \quad (1)$$

where  $V$  = the volume of the liquid permeated through the GCL,  $t$  = time,  $c_0$  = the source concentration of the salt cation,  $i$  = the hydraulic gradient,  $A$  = the cross-sectional area of flow, and  $k(t)$  = the hydraulic conductivity which, as shown in Fig. 7, is a function of time due to time-dependent swell and interactions between the bentonite and the chemical solution. For a constant hydraulic gradient, the volumetric flux of chemical solution entering the GCL can be evaluated as follows:

$$\int_0^V \frac{dV}{A} = i \cdot \int_0^t k(t) \cdot dt \quad (2)$$

or

$$\frac{V}{A} = i \cdot \int_0^t k(t) \cdot dt \quad (3)$$

On the basis that chemical equilibrium occurs when the amount of salt delivered to the GCL in the field is equivalent to that delivered to the GCL specimen in the laboratory, we can write:

$$i_F \cdot \int_0^{t_F} k_F(t) \cdot dt = i_L \cdot \int_0^{t_L} k_L(t) \cdot dt \quad (4)$$

or

$$\int_0^{t_F} k_F(t) \cdot dt = \frac{i_L}{i_F} \cdot \int_0^{t_L} k_L(t) \cdot dt \quad (5)$$

where the subscript 'F' refers to the field and the subscript 'L' refers to the laboratory.

In order to evaluate Eq. 5, the functional relationships between the hydraulic conductivities in the laboratory and the field with time are required. Although a simplified functional relationship may be possible in the case of the laboratory test data, the functional relationship in the field is unknown *a priori*. However, one approach is to assume that the temporal trend in the hydraulic conductivity in the field is the same as that in the laboratory (i.e., for the same initial and boundary conditions), except the trend is extended over a longer duration due to a lower hydraulic gradient in the field relative to that in the laboratory.

For example, consider the laboratory hydraulic conductivity data in Fig. 7 for the non-prehydrated GCL specimen permeated with a 20-mM  $\text{CaCl}_2$  solution. As shown in Fig. 9, the trend in this data [i.e.,  $k_L(t)$ ] can be reasonably approximated by fitting the data with a 6<sup>th</sup>-order polynomial function, resulting in a coefficient of determination,  $r^2$ , of 0.972. The time required to achieve chemical equilibrium in this test ( $t_L$ ) was 164 days under an applied hydraulic gradient ( $i_L$ ) of approximately 200. If the temporal trend in hydraulic conductivity in the field is the same as that in the laboratory [i.e.,  $k_F(t) = k_L(t)$ ], then the time required to achieve chemical equilibrium and, therefore, steady-state hydraulic conductivity in the field can be determined as a function of the hydraulic gradient in the field ( $i_F$ ) by finding the root,  $t_F$ , of  $k_F(t)$  that satisfies the left-hand side of Eq. 5.

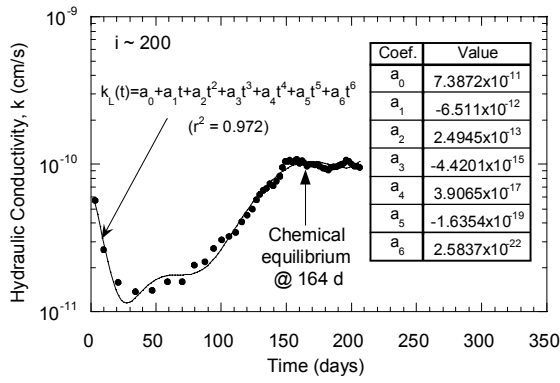


Figure 9. Measured and fitted temporal trends in hydraulic conductivity data for a non-prehydrated GCL permeated with a 20-mM  $\text{CaCl}_2$  solution.

Analysis of the data in Fig. 9 using this procedure results in the correlation between  $t_F$  and  $i_F$  shown in Fig. 10. As indicated by the results in Fig. 10, substantially lower hydraulic gradients in the field can result in substantially longer times required to achieve chemical equilibrium in the field relative to those in the laboratory, such that the overall long-term hydraulic performance of the GCL may not be realized for tens of years. Of course, the results of this analysis are oversimplified due to the assumptions of a constant source concentration and constant hydraulic gradient. However, the example serves to illustrate the need for continuous monitoring of the performance of waste containment systems over the long term.

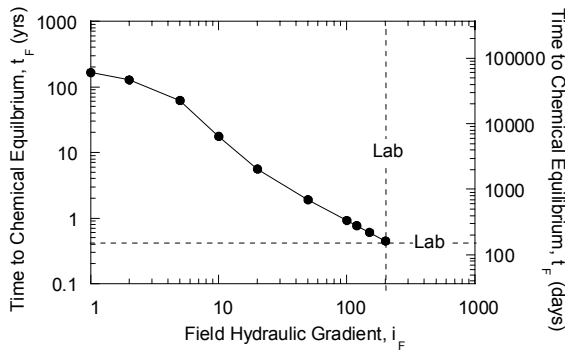


Figure 10. Estimated correlation between times to achieve chemical equilibrium and hydraulic gradient in the field based on the results of laboratory test data for a non-prehydrated GCL permeated in the laboratory with a 20-mM  $\text{CaCl}_2$  solution under a hydraulic gradient of approximately 200 (see data in Fig. 9).

All of the test results shown in Figs. 7 and 8 are based on permeation of GCLs with single-salt solutions (i.e.,  $\text{CaCl}_2$ ) containing a single, divalent cation ( $\text{Ca}^{2+}$ ). Thus, the question arises as to how representative are the results in terms of actual practical applications wherein the containment liquids (e.g., leachate) include a multitude of different cations at different concentrations? This question was recently addressed by Kolstad (2000) and Kolstad et al. (2004) in terms of the correlation between hydraulic conductivity and two parameters characterizing the chemistry of multi-species inorganic solutions, viz. the ionic strength,  $I$ , and the ratio of the concentrations of the monovalent to the divalent cations in the permeant solution, RMD, where:

$$I = \frac{1}{2} \sum_{i=1}^N c_i z_i^2 \quad (6)$$

and

$$\text{RMD} = \frac{M_M}{\sqrt{M_D}} \quad (7)$$

where  $c$  = the molar concentration of each species (both anions and cations) in solution,  $z$  = the valence or charge of the species,  $N$  = the number of ions in solution,  $M_M$  = the total molarity of monovalent cations, and  $M_D$  = the total molarity of divalent cations. The resulting correlation based on numerous leachates from municipal solid wastes (MSW), construction and demolition (C&D) waste, fly ash, and mine waste is shown in Fig. 11. As indicated in Fig. 11, the predicted hydraulic conductivity of GCLs increases with increasing  $I$  for a given RMD as well as decreasing RMD for a given  $I$ .

To provide an example of the accuracy of the predictions of hydraulic conductivity based on Fig. 11, the measured hydraulic conductivities for the non-prehydrated GCLs in Fig. 7 are compared in Fig. 12 with the predicted values using Fig. 11 based on the various ionic strengths for the  $\text{CaCl}_2$  solutions and RMD = 0 (i.e., no monovalent cations in solution). As shown in Fig. 12, the predicted hydraulic conductivities, with the exception of one result, are all somewhat higher than the measured values, suggesting that predicted hydraulic conductivities based on  $I$  and RMD will be, at worst, somewhat conservative (i.e., too high).

Based on the data shown in Fig. 11, Kolstad et al. (2004) concluded that hydraulic conductivities of GCLs  $> 10^{-7}$  cm/s should not be common in bottom liners where leachates similar to those used in the evaluation are likely to be found, and that many of the data associated with higher hydraulic conductivities correspond to leachates from 'young' MSW landfills ( $< 5$  yr).

However, Kolstad et al. (2004) also note that the composition of MSW leachate changes over time such that high hydraulic conductivities may not be realized for some time because of the relatively long time required for a GCL and leachate to reach equilibrium under field conditions.

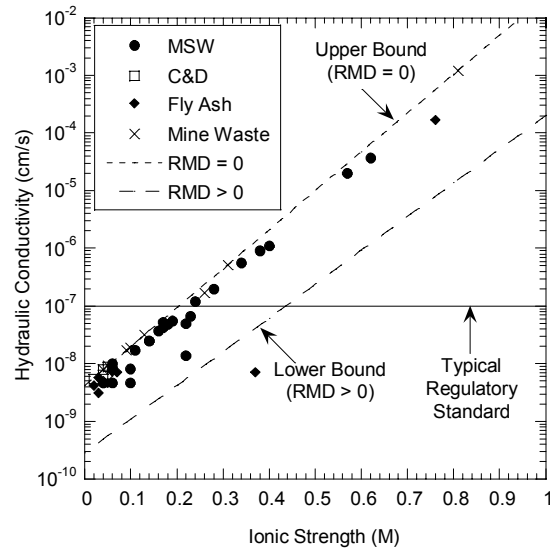


Figure 11. Predicted correlation between hydraulic conductivity of non-prehydrated GCLs and the ionic strength ( $I$ ) and ratio of concentrations of monovalent to divalent cations (RMD) in solution for a wide variety of actual leachates (MSW=municipal solid waste; C&D = construction and demolition waste) (replotted from Kolstad, 2000).

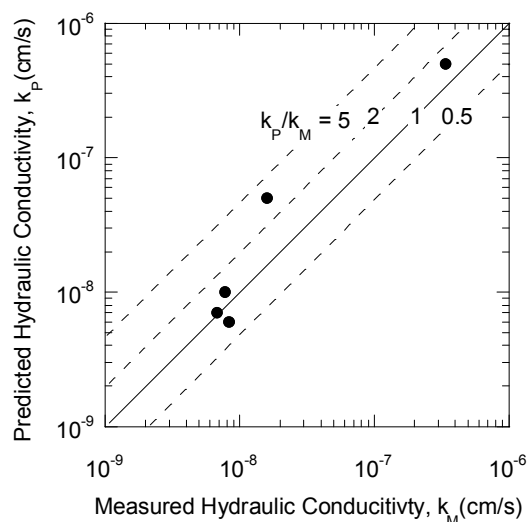


Figure 12. Measured versus predicted hydraulic conductivities for specimens of a non-prehydrated GCL permeated with calcium chloride solutions.

### 3 ALTERNATIVE BARRIERS AND BARRIER MATERIALS

Alternative barriers can be defined as barriers, such as liners and covers for waste containment systems, that are used as alternatives to regulated or prescribed barriers, such as those shown in Fig. 5 and summarized by Koerner et al. (1998) and Koerner and Koerner (1999), but have equivalent or better performance relative to the regulated barriers. Alternative barrier materials refer to materials used as alternative barriers or as components of alternative barriers, and may include non-soil or non-polymer materials, such as asphalt (Bowders et al., 2000; 2002, 2003; Neupane et al., 2005a,b) and paper mill sludge (Moo-Young and Zimmie, 1996, 1997; Moo-Young et al., 2000; Ochola and Moo-Young, 2004), or unusual soils that may not be capable of achieving typically prescribed properties (e.g.,  $k \leq 10^{-7}$  cm/s), such as some residual or lateritic soils (Frempong and Yanful, 2005; Krisdani et al., 2005; Oliveira Filho et al., 2005; Osinubi and Nwaiwu, 2005) and some marine deposits (Kamon and Katsumi, 2001; Kamon et al., 2002; Du and Hayashi, 2005). Consideration of alternative barriers and barrier materials usually occurs when

- issues of cost become paramount,
- prescribed materials are not readily available, and/or
- the concern for the potential consequences of poor barrier performance exceeds that typically required or assumed.

In terms of covers, interest in the use of alternative earthen final covers (AEFCs) relative to regulated or prescriptive covers (i.e., composite or compacted clay covers) has been gaining due to the relatively high costs typically associated with prescriptive covers (~\$400,000US to \$500,000US/ha) and the failure of some prescriptive covers constructed with compacted clay in regions with drier climates (e.g., due to desiccation) (e.g., Dwyer, 1997, 2001). Alternative earthen final covers are alternative covers comprised entirely of soils and designed on the basis of water balance principles to perform as well as, if not better than, the prescriptive counterparts and typically with greater durability and/or lower cost.

As shown in Fig. 13, the basic concept for AEFCs is to provide a soil water storage capacity within the AEFC that is greater than that needed to store water when the evapotranspiration (ET) component of the water balance is low, typically dur-

ing the winter (Stormont and Morris, 1998; Khire et al., 2000). Alternative earthen final covers usually are suitable in drier regions where potential evapotranspiration (PET) significantly exceeds precipitation (P) (i.e.,  $PET > 2P$ ), although AEFCs also may be built in wetter climates. Although a variety of design concepts and terminology have been used for AEFCs, the two most common types of AEFCs are monolithic covers (e.g., Zornberg et al., 2003; Albright et al., 2004; Benson et al., 2004, 2005; Oliveira Filho et al., 2005; Somasundaram et al., 2005) and capillary barrier covers (e.g., Aubertin et al., 1994; Khire et al., 1994; Benson and Khire, 1995; Stormont 1995; Stormont et al., 1996; Watkins, 1996; Morris and Stormont, 1997; Stormont and Morris, 1998; Stormont and Anderson, 1999; Albright et al., 2004; Krisdani et al., 2005; Parent and Cabral, 2005).

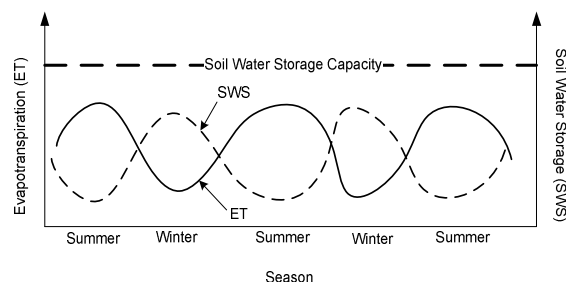


Figure 13. Concept of soil-water storage capacity for alternative earthen finer covers (after Stormont and Morris, 1998; Khire et al., 2000).

As illustrated conceptually in Fig. 14, monolithic covers or MCs (also known as monocovers, store-and-release covers, soil-plant covers, evapotranspirative covers, or phytocovers) consist of a relatively thick, single layer of comparatively fine-textured soil with a relatively high water storage capacity. On the other hand, a capillary barrier cover (CBC) is a two-layered system consisting of a relatively fine textured soil overlying a relatively coarse textured soil (see Fig. 14). A CBC is based on the concept of the capillary break that occurs between the finer textured soil and the coarser textured soil under unsaturated soil conditions. For CBCs, as long as the suction at the interface of the two layers ( $\psi$ ) is higher than the suction at the intersection of the curves in hydraulic conductivity versus soil suction for the two soils (i.e.,  $\psi > \psi_i$ ), the unsaturated hydraulic conductivity in the coarser textured soil will be lower than that for the finer textured soil such that flow of infiltrating water into the coarser textured soil will be impeded.

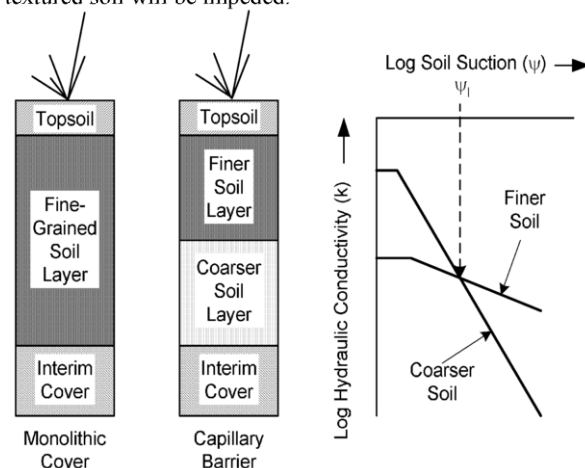


Figure 14. Typical cross sections of the two main types of alternative earthen final covers.

Failure of a CBC will occur when the soil suction at the interface between the layers reaches a value corresponding to the sharp bend in the soil water characteristic curve (SWCC) of the coarser soil near residual water content (Khire et al., 1999).

Thus, a CBC generally is considered plausible only in regions with relatively small precipitation events, such as in arid and semi-arid climates. However, even in arid and semi-arid regions, provision must be made for adequate lateral drainage of infiltrating water to minimize the potential for saturation of the finer layer, particularly when the finer layer is relatively thin.

The advantages of AEFCs such as MCs and CBCs relative to more traditional, prescribed cover systems with one or more resistive layers with low saturated hydraulic conductivities are as follows:

- (1) AEFCs typically are less susceptible to desiccation and cracking since AEFCs can be constructed using relatively non-plastic soils [e.g., low plasticity silts (ML) and/or low plasticity silty sands (SM)];
- (2) AEFCs are relatively simple to construct since high soil-water storage capacity is more important than the use of heavy compaction to achieve a low saturated hydraulic conductivity;
- (3) AEFCs are more economical since AEFCs can be built using a wide range of soil types, usually within the local vicinity of the site; and
- (4) AEFCs require relatively low post-closure maintenance.

The primary advantage of a CBC relative to an MC is that the capillary break in a CBC essentially increases the soil water storage capacity of the overlying finer textured soil relative to that which would exist in the case of an MC. However, because of the requirement for two, contrasting soil layers in a CBC, CBCs often are more expensive than MCs. Considerations for the design of AEFCs can be found in Morris and Stormont (1997), Stormont and Morris (1998), Khire et al. (2000), Benson and Chen (2003), Zornberg et al. (2003), and Parent and Cabral (2005).

The key requirement for regulatory acceptance of an AEFC is that the AEFC must have, as a minimum, equivalent performance relative to that for the prescribed cover systems. In this regard, the U. S. EPA is conducting an assessment of the performance of AEFCs relative to that for prescribed covers through the Alternative Cover Assessment Program, or ACAP (Albright et al., 2004). The ACAP consists of the monitoring of relatively large (10 x 20 m), fully instrumented drainage lysimeters with cross-sections representing conventional covers employing resistive barriers or alternative covers (MCs and CBCs) constructed at 11 field sites in the United States. The locations of the field sites, shown in Fig. 15, include wet and dry climates as well as warm and cold climates. Based on the ratio of precipitation (P) to potential evapotranspiration (PET), Albright et al. (2004) indicate that one site is located in an arid climate ( $0.03 < P/PET \leq 0.2$ ), six sites are located in semiarid climates ( $0.2 < P/PET \leq 0.5$ ), one site is located in a sub-humid climate ( $0.5 < P/PET \leq 0.75$ ), and three sites are located in humid climates ( $P/PET > 0.75$ ).

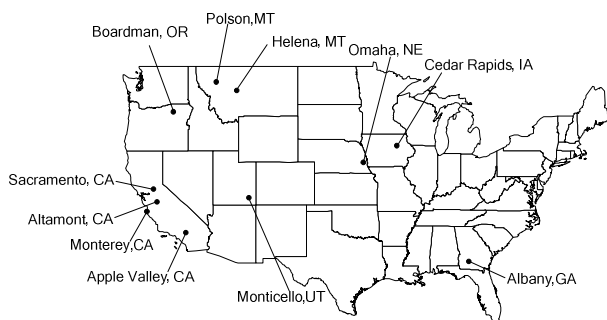


Figure 15. Locations of the sites in the United States where large-scale covers have been constructed and are being monitored as part of the U.S. EPA's Alternative Cover Assessment Program (from Albright et al., 2004).

A schematic of the cross section of the drainage lysimeters is shown in Fig. 16. Surface runoff (SRO) and percolation ( $P_r$ ) are monitored via collection from the top and base of the cover, respectively. Water content is monitored with low-frequency time domain reflectometry (TDR) probes and matric suction is monitored with thermal dissipation sensors. A weather station is used for meteorological monitoring. Soil-water storage (SWS) is computed by integrating the distribution of water content. Evapotranspiration (ET) is computed as the residual of the water balance as follows (Benson et al., 2004):

$$ET = P - SRO - P_r - \Delta(SWS) \quad (8)$$

The ET computed with Eq. 8 includes actual ET and the net error in the other water balance quantities.

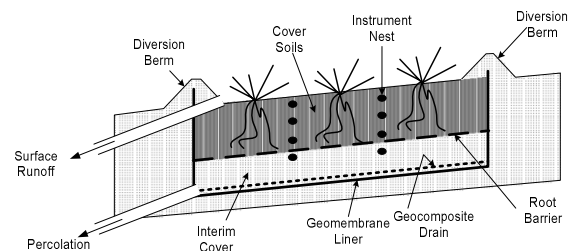


Figure 16. Schematic cross section of cover lysimeters used for measuring water balance of alternative covers for the U.S. EPA's Alternative Cover Assessment Program (redrawn after Albright et al., 2004).

Examples of the some of the measured data for three AEFCs at two sites located in semiarid climates (Altamont, CA, and Boardman, OR) and one site located in a sub-humid climate (Polson, MT) are shown in Fig. 17. Based on the data shown in Fig. 17, some general observations can be made. First, keeping the scales in mind, all three AEFCs have performed extremely well in terms of limiting percolation, with essentially no percolation being measured at the Boardman and Polson sites, and only very low percolation rates ( $< 1.5$  mm/yr, on average) at the Altamont site. Albright et al. (2004) also report that very low percolation rates ( $< 1.5$  mm/yr and 0.2 % of precipitation, on average) have been recorded for 7 of the 10 AEFCs that are located at sites in arid, semiarid, and sub-humid climates. Second, aside from the initial period after construction, when SWS is relatively high during the initial monitoring stage, the greatest amount of water removed from the AEFCs is attributed to evapotranspiration (ET), underscoring the importance of vegetation, whereas the least amount of water removed from the AEFCs is due to surface runoff (SRO). Albright et al. (2004) report that, in general, SRO has been only a small fraction of the water balance, ranging from 0.0 to 10.2 % of precipitation with an average of only 3.8 %, and that the mean SRO reported as a fraction of precipitation is statistically independent of slope (5 % vs. 25 %), type of cover (prescribed vs. AEFC), and climate (arid vs. humid). Third, except towards the end of the record for the monolithic cover at Altamont, where SWS remains relatively high during the Spring and Summer of 2003, the seasonal trend expected in SWS shown schematically in Fig. 17 tends to be mimicked well. In the case of Altamont, Benson et al. (2004) note that approximately twice as much precipitation was received in the Winter of 2003 than in previous years such that the transpiration capacity of the plants was exceeded allowing the soil water to accumulate, and that phenology of the vegetation also appeared to have been altered such that transpiration ceased much earlier than normal (hence, the lack of a drop in SWS during the Spring and Summer of 2003). This observation again underscores the importance of the vegetation in terms of the performance of AEFCs. Finally, periodic jumps in P (indicated

by vertical arrows) at each of the three sites correspond well with similar jumps in SWS, implying a direct correlation between P and SWS, as expected.

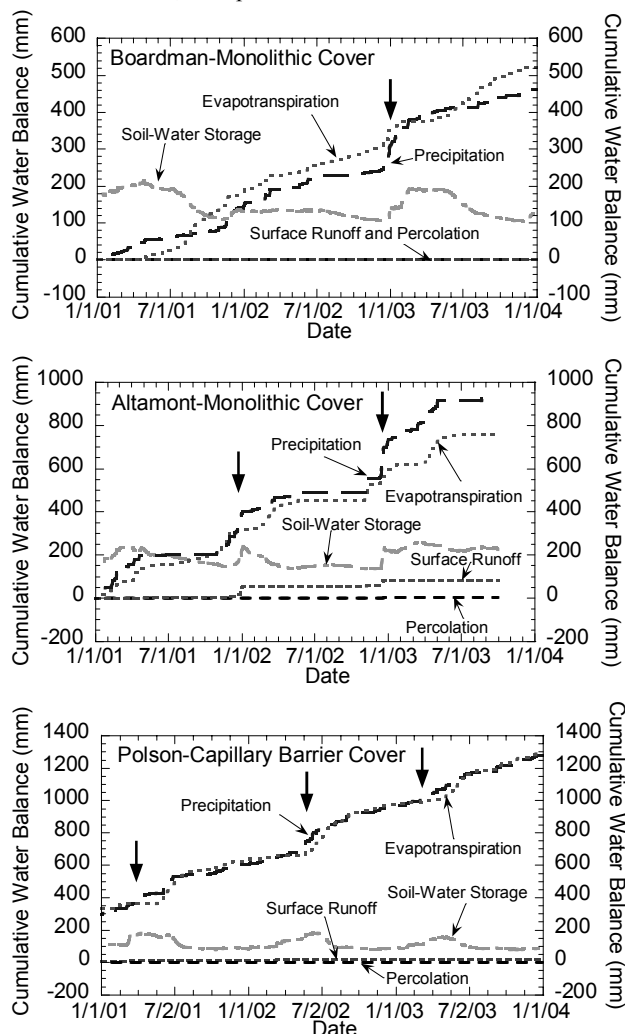


Figure 17. Measured water balance data for alternative earthen covers at three sites in the U.S. EPA's Alternative Cover Assessment Program.

Based on the results of the ACAP to date, where the performance of prescriptive (regulated or conventional) covers is being assessed alongside that of AEFCs, maximum percolation rates for AEFCs that provide equivalent hydrologic performance have been established. As shown in Table 1, these maximum percolation rates are a function of the climate and the type of prescriptive cover that the AEFC is to replace. However, further monitoring of both conventional covers and AEFCs at a variety of locations and climates is required either to confirm the criteria given in Table 1, or to adjust the criteria as necessary based on further inputs.

Table 1. Equivalent percolation rates for alternative earthen final covers relative to prescriptive final covers based on the U.S. EPA's Alternative Cover Assessment Program (data from Albright et al., 2004).

| Type of Prescriptive Cover | Maximum Annual Percolation (mm/yr) <sup>(1)</sup> |                      |
|----------------------------|---------------------------------------------------|----------------------|
|                            | Arid, Semiarid, and Sub-humid (P/PET ≤ 0.75)      | Humid (P/PET > 0.75) |
| Compacted Clay (or lesser) | 10                                                | 30                   |
| Composite                  | 3                                                 | 3                    |

<sup>(1)</sup> P = precipitation; PET = potential evapotranspiration

#### 4 INNOVATIVE BARRIERS AND BARRIER MATERIALS

Innovative barriers and barrier materials are distinguished from alternative barriers and barrier materials in that the former are still in the fundamental stage of research and development whereas the latter are more in the demonstration stage for regulatory approval and application. The objective in developing innovative barriers and barrier materials is to produce a barrier that is more efficient and/or less costly than existing barriers. Improved efficiency in this case refers to better performance in terms of containment or sustainability of containment. Although there are a wide variety of possible innovative barriers and barrier materials (e.g., see Lo, 2003; Kaya and Durukan, 2004; Darwish et al., 2004), the focus of the current presentation will be limited to two types of innovative barriers and barrier materials, viz. (1) clay membrane barriers, and (2) polymer-clay nanocomposites.

##### 4.1 Clay membrane barriers

A significant amount of recent research has focused on the potential benefits arising from the existence of semi-permeable membrane behavior in bentonites and/or bentonite-based barrier materials (Malusis et al., 2001; Malusis and Shackelford 2002a,b; Van Impe, 2002; Manassero and Dominijanni, 2003; Shackelford et al., 2003; Henning, 2004; Lu et al., 2004; Dominijanni and Manassero, 2005; Yeo et al., 2005). The existence of membrane behavior in clays is evident by restricted passage of solutes as well as by chemico-osmosis, or the movement of liquid from lower solute concentration (higher water activity) to higher solute concentration (lower water activity). Restricted movement of charged solutes (ions) through the pores of a clay soil is attributed to electrostatic repulsion of the ions by electrical fields associated with the adsorbed layers of ions (commonly referred to as diffuse double layers or DDLs) of adjacent clay particles. Non-charged solutes, such as some aqueous miscible organic compounds comprised of long carbon chains, are restricted when the size of the solute species is greater than the pore size available for migration.

Clay membrane behavior is quantified in terms of a chemico-osmotic efficiency coefficient,  $\omega$ , also commonly known as a reflection coefficient,  $\sigma$ . The value of  $\omega$  for a clay soil exhibiting no solute restriction is zero ( $\omega = 0$ ), corresponding to zero membrane efficiency, whereas the value of  $\omega$  for a clay soil exhibiting complete solute restriction is unity ( $\omega = 1$ ), corresponding to 100 percent membrane efficiency. In general, the values of  $\omega$  for naturally occurring clay soils that exhibit membrane behavior range from greater than zero to less than unity (i.e.,  $0 < \omega < 1$ ) because of the variation in pore sizes that exist in such soils. Membranes that are 100 percent efficient are referred to as "ideal" or "perfect" membranes. All materials that exhibit membrane behavior are called "semi-permeable" membranes, since all membranes are permeable to the solvent (water) regardless of the efficiency of solute restriction.

Clay membrane behavior is a function of several mechanical, physical, and chemical factors, including the stress-strain properties of the clay, the boundary and initial salt concentrations, the types of solute species (ions), and the mineralogy of the soil (Shackelford et al., 2003). In general, the potential for the existence of membrane behavior increases with (a) an increase in stress (decrease in porosity), (b) an increase in content of high activity clay minerals, particularly sodium montmorillonite, and (c) a decrease in the salt concentration in the pore water (Shackelford et al., 2003).

Three mechanisms contribute to the beneficial aspects resulting from the existence of membrane behavior in clay barriers, viz., (1) hyperfiltration, (2) chemico-osmotic flow, and (3) reduced diffusive mass transport (Malusis, 2001; Malusis and

Shackelford, 2002c; Shackelford et al., 2003; Malusis and Shackelford, 2004a,b). The hyperfiltrated advective flux represents the traditional advective transport term that is reduced by a factor of  $(1 - \omega)$  due to the membrane behavior of the soil. In physical terms, the factor  $(1 - \omega)$  is considered to represent the process of hyperfiltration whereby solutes are filtered out of solution as the solution passes through the membrane under an applied hydraulic gradient. Chemico-osmotic flow results from the establishment of a concentration gradient across the barrier due to solute restriction such that water flows in the direction of increasing solute concentration (i.e., decreasing water activity). For typical containment barrier scenarios, chemico-osmotic flow would be directed towards the side of containment thereby reducing the net outward flux of the contaminants. The reduced diffusive mass transport results from the decrease in the effective solute diffusion coefficient,  $D^*$ , with increasing  $\omega$  such that in the limit as  $\omega \rightarrow 1$ ,  $D^* \rightarrow 0$ , i.e., because no solute can migrate through an ideal ( $\omega = 1$ ) semi-permeable membrane (Malusis and Shackelford, 2002b).

The potential significance of membrane behavior is illustrated in Fig. 18, where the ratio of solute mass flux at steady state through a 1-m-thick clay barrier that behaves as a semi-permeable membrane ( $J_m$ ) to that which exists for non-membrane behavior ( $J_{nm}$ ) is plotted as a function of  $\omega$  and the hydraulic gradient ( $i$ ). The results shown in Fig. 18 are based on simulations using a coupled solute transport model and measured values for  $D^*$  and  $\omega$  as described by Malusis (2001) and Malusis and Shackelford (2004a). As shown in Fig. 18, in the absence of membrane behavior,  $\omega = 0$  such that  $J_m = J_{nm}$ . However, as membrane behavior becomes more significant (i.e., as  $\omega$  increases), the solute mass flux exiting the barrier is increasingly reduced such that  $J_m < J_{nm}$ . For example, at a membrane efficiency of 60 % ( $\omega = 0.6$ ), the exit solute mass flux at steady state is only about 30 % of that which would exist in the absence of membrane behavior (i.e.,  $J_m/J_{nm} \approx 0.3$ ). In the limit as  $\omega \rightarrow 1$ ,  $J_m/J_{nm} \rightarrow 0$  because, by definition, there can be no solute mass transport through an ideal or perfect membrane. The results for the two cases where a hydraulic gradient is applied (i.e.,  $i = 10$  and  $i = 100$ ) are essentially the same as the results for the pure diffusion case ( $i = 0$ ) because transport through the barrier is controlled by diffusion due to the very low hydraulic conductivity of the barrier material ( $k \approx 10^{-9}$  cm/s). As a result, the hyperfiltration and chemico-osmotic mechanisms of membrane behavior are essentially negligible in this case.

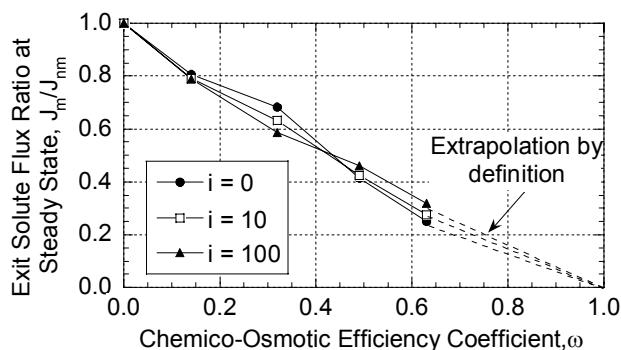


Figure 18. Effect of membrane behavior on solute mass flux at steady state through a 1-m-thick clay barrier ( $i$  = hydraulic gradient;  $J_m$  = membrane steady-state solute flux;  $J_{nm}$  = non-membrane steady-state solute flux) (results from Malusis, 2001; Malusis and Shackelford, 2004a).

The existence of membrane behavior in sodium bentonite is fortuitous in that several types of geoenvironmental containment barriers consist of either sodium bentonite, such as geosynthetic clay liners (GCLs), or soil mixtures containing sodium bentonite, such as compacted sand-bentonite liners and soil-bentonite (SB) vertical cutoff walls. Thus, the potential for membrane behavior resulting from the sodium bentonite content in clay soil barrier materials is high (Shackelford et al., 2003).

For example, consider the results in Fig. 19 for two model soil-bentonite (SB) backfill mixtures consistent with those commonly used in SB vertical cutoff walls. The base soils for the backfills consisted of natural clay (Nelson Farm Clay) with 89 % (w/w) low plasticity ( $PI = 14.5$ ) fines and a mixture of sand with 5 % (w/w) dry sodium bentonite ( $PI = 454$ ). Specimens of both base soils were mixed with a sufficient amount of 5 % (w/w) sodium bentonite-water slurry to correspond to a 100-mm slump in accordance with standard practice for SB vertical cutoff walls. The membrane behavior was evaluated by measuring the  $\omega$  resulting from maintaining a 3.88-mM KCl concentration difference across the specimen. After a steady-state  $\omega$  was achieved for the specimen at an initial void ratio ( $e$ ), the specimen was compressed to a new, lower void ratio until a new steady-state  $\omega$  was achieved. This procedure was repeated such that a total of three  $\omega$  values were recorded for each SB backfill mixture.

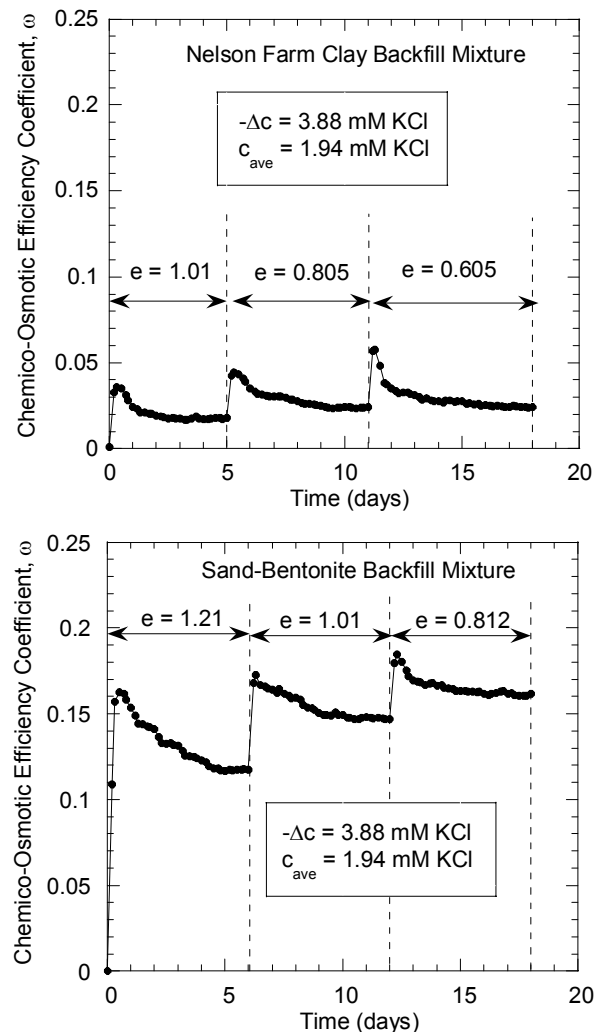


Figure 19. Temporal trends in chemico-osmotic efficiency as a function of void ratio ( $e$ ) for two soil-bentonite backfills (data from Yeo, 2003; and Yeo et al., 2005).

The results in Fig. 19 indicate that both backfills act as membranes, with  $\omega$  ranging from 0.018 to 0.024 for the natural clay backfill mixture and from 0.118 to 0.166 for the sand-bentonite backfill mixture. The higher  $\omega$  values for the sand-bentonite backfill mixture illustrate the significance of the mineralogy of the clay portion of the backfill mixtures. Also, more significant membrane behavior (higher  $\omega$ ) is correlated with lower void ratio (higher consolidation stress), as expected.

In terms of real-world SB backfills, two SB backfills recovered from cutoff walls constructed in Delaware and New Jersey were recently tested for the existence of membrane behavior (Henning, 2004). Both SB backfills were mixed with 3 to 4 % bentonite by dry weight with the locally excavated soil. Both backfills were classified as sands (i.e., SC and SP-SC) with the fines portions classified as medium plastic clays (CL). The results, shown in Fig. 20, indicate that both backfills exhibited membrane behavior, although the magnitude of the membrane behavior is lower than that for the model backfills prepared in the laboratory (Fig. 19). The difference in magnitudes is attributed, in part, to lower percentages of clay in the actual backfills relative to the model backfills. Nevertheless, calculations by Henning (2004) indicate that the total liquid flux of cutoff walls comprised of the two actual backfills would be reduced from 8 to 13 %, depending on the backfill, due to the existence of the membrane behavior.

One of the issues related to CMBs is sustainability of the membrane behavior (Shackelford et al., 2003). For example, recently reported results indicate that there is a direct correlation between diffusion of invading cations into bentonites and either partial or complete destruction of membrane behavior (Malusis and Shackelford, 2002a; Shackelford and Lee, 2003). The data shown in Fig. 21 illustrate how the chemico-osmotic pressure difference (i.e., the pressure driving water through a membrane) varies with valence of the invading cation. The data are from chemico-osmotic tests conducted with specimens of bentonite subjected to KCl (20 mM) and CaCl<sub>2</sub> (5 mM) solutions. A concentration difference was imposed by circulating the salt solution across the top of the specimen while simultaneously circulating deionized water across the bottom of the specimen. A closed system was used where liquid flow was prevented so that the build up in osmotic pressure ( $\Delta P$ ) could be observed (e.g., see Malusis et al., 2001). As shown in Fig. 21,  $\Delta P$  rises immediately for both solutions in response to the concentration difference. Subsequently,  $\Delta P$  reaches a stable value (K<sup>+</sup> test) or rapidly diminishes (Ca<sup>2+</sup> test). A stable  $\Delta P$  is achieved with the monovalent cation because exchange of K<sup>+</sup> for Na<sup>+</sup> does not appreciably affect osmotic swelling of the montmorillonite, which is responsible for membrane behavior. In contrast,  $\Delta P$  diminishes for the divalent cation because exchange of Ca<sup>2+</sup> for Na<sup>+</sup> eliminates osmotic swelling of the montmorillonite layers, effectively destroying the membrane behavior.

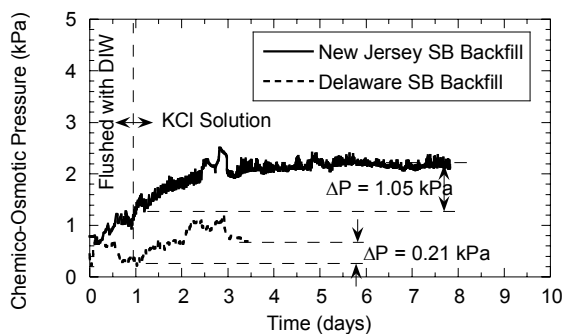


Figure 20. Chemico-osmotic pressure buildup resulting from establishing and maintaining an 8.7-mM KCl concentration difference across two real-world SB backfills recovered from the field and tested in the laboratory (data from Henning, 2004).

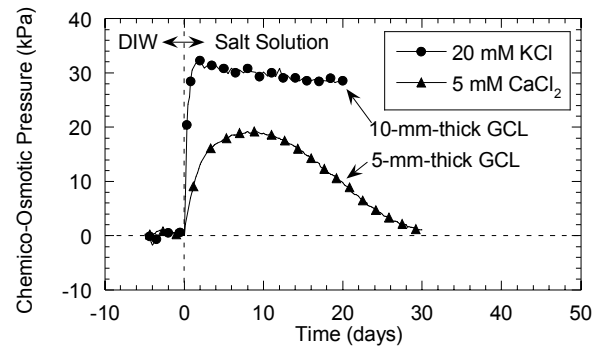


Figure 21. Temporal trends in chemico-osmotic pressure difference for monovalent (K<sup>+</sup>) and divalent (Ca<sup>2+</sup>) cations invading the bentonite contained in two geosynthetic clay liners (GCLs) during chemico-osmotic membrane tests (data from Malusis and Shackelford, 2002b; Shackelford and Lee, 2003).

The effect of partial or complete destruction of membrane behavior due to diffusion of invading cations illustrated in Fig. 21 is similar to the effect of permeation with salt solutions on the hydraulic conductivity of GCLs as previously discussed (see Fig. 7). That is, permeation with salt solutions can collapse the structure of the bentonite particles resulting in the formation of larger inter-aggregate pores and an increase in hydraulic conductivity. Thus, an important issue that warrants further attention is how to alter or otherwise modify the composition or structure of bentonite such that the adsorbed layer of cations associated with the bentonite particles can resist collapse and the existence of any membrane behavior can be sustained.

#### 4.2 Polymer-clay nanocomposites

Polymer-clay nanocomposites, where the clay specifically pertains to the layered silicates, are being used in a variety of industries because of superior mechanical, thermal, and electrical properties relative to the polymer alone. For example, polymer-clay nanocomposites have been used as containers due to lower gaseous diffusion coefficients, as bumpers and dashboards in the automotive industry due to greater strength and ductility, and in the electronics and computer industry because of superior thermal and electrical properties (Kornmann et al., 1998).

In general, polymer-clay nanocomposites consist of plastic materials that contain a small fraction of layered silicates (usually less than a few percent by weight) dispersed within the polymer matrix. The resulting composites are referred to as nanocomposites because the change in composition and structure resulting from addition of the dispersed layered silicates occurs over a nanometer length in scale. Such polymer-clay nanocomposites have been formed using a wide range of polymers, including epoxy, polyurethane, polypropylene, polyimide, polystyrene, rubber (nitrile), poly( $\epsilon$ -caprolactone), and polysiloxane (LeBaron et al., 1999).

Polymer-clay nanocomposites are formed by dispersing or "mixing" layered silicate clay minerals, primarily the smectites, within a polymer matrix. Layered silicate nanolayers are preferred because the high aspect ratio of layered silicates (i.e., ratio of the length to thickness of individual clay mineral layers) is ideal for reinforcement. However, dispersion of layered silicates as separate monolayers into polymers is not readily accomplished for two reasons. First, layered silicates tend to prefer a face-to-face stacking that results in agglomerated tactoids that are not readily separable. Second, the hydrophilic nature of layered silicates is intrinsically incompatible with the inherent hydrophobicity of polymers used in engineering applications. These difficulties can be overcome by first converting the layered silicates into organoclays by replacing the inorganic exchange cations in the interlayers or galleries of the native clay with alkylammonium surfactants or quaternary ammonium ions

(i.e., onium ions). The degree of this conversion and extent to which the replacement is successful in separating the layered silicates into separate monolayers results in three types of polymer-layered silicate nanocomposites, as illustrated schematically in Fig. 22.

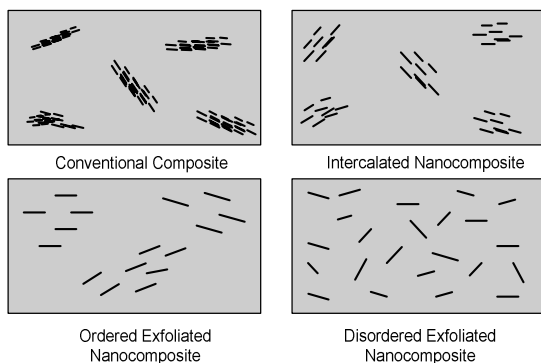


Figure 22. Schematic illustrations of compositions of different types of nanocomposites (redrawn after LeBaron et al., 1999).

Conventional or entrained polymer-clay composites are formed by mixing naturally occurring (i.e., unconverted) powdered layered silicates, usually montmorillonite in the form of bentonite, with a liquid polymer. The resulting composites contain aggregated nanolayer tactoids that ordinarily improve rigidity, but often lack sufficient improvement in strength, elongation and toughness (see Fig. 22). Nanocomposites are referred to as intercalated composites if the replacement of inorganic exchangeable cations with alkylammonium surfactants results in layers that persist with a repeating stacking pattern such that the interlayer or gallery heights are less than two times the onium ion chain length. In this case, regions of the composite will consist of very high and very low reinforcer concentrations (see Fig. 22). For true nanocomposites, the clay nanolayers must be uniformly dispersed or exfoliated as monolayers existing within the polymer matrix. Such ideal nanocomposites are formed upon polymerization only when the clay layers are forced apart and no longer interact through the onium ion chains. The difference between the ordered and disordered exfoliated nanocomposites illustrated in Fig. 22 is that the former can be detected by X-ray diffraction and the latter is X-ray amorphous (LeBaron et al., 1999).

As previously mentioned, polymer-clay nanocomposites possess engineering properties that typically are substantially improved relative to the properties of the polymer alone. For example, consider the stress-strain data shown in Fig. 23 for a linear low-density polyethylene (LLDPE, Dowlex® 2032, MI = 2.0 g/10 min, density = 0.926 g/cc) and polymer-clay nanocomposites comprised of dimethylbis(hydrogenated-tallow) ammonium montmorillonite [ $M_2(HT)_2$ ] dispersed in the LLDPE at different montmorillonite (MMT) contents by dry weight. The addition of only 2.5 % MMT as the organoclay improves the strength of the LLDPE by almost 50 %, whereas as little as 6.9 % MMT results in almost double the strength of the LLDPE.

Another effect resulting from dispersion of layered silicates is a significant decrease in the gas permeability relative to that of the polymer. For example, as shown in Fig. 24, the permeability of LLDPE to oxygen ( $O_2$ ), nitrogen ( $N_2$ ) and carbon dioxide ( $CO_2$ ) is significantly reduced with increasing MMT content in the form of the  $M_2(HT)_2$  organoclay dispersed in the LLDPE. The reduction in gas permeability is attributed to the increased tortuosity for gas migration through the polymer due to the presence of the exfoliated monolayers of the layered silicates, as

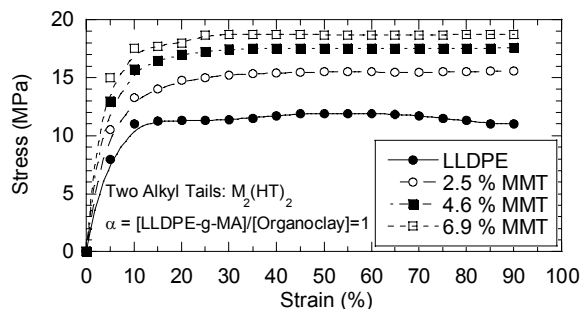


Figure 23. Stress-strain curves for linear low density polyethylene (LLDPE) and LLDPE-organoclay nanocomposites with different weight percentages of montmorillonite (MMT) (data from Hotta and Paul, 2004).

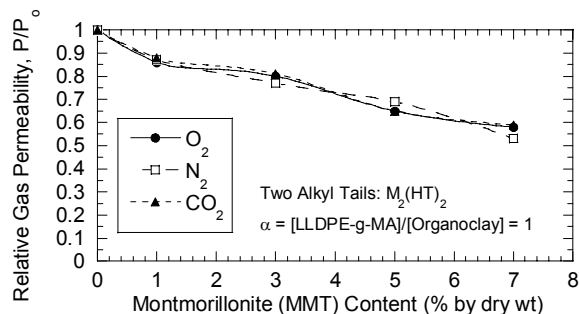


Figure 24. Relative permeability of oxygen, nitrogen, and carbon dioxide gases for 150- $\mu$ m-thick organoclay nanocomposite films with different montmorillonite contents at 35 °C (data from Hotta and Paul, 2004).

illustrated schematically in Fig. 25 (Yano et al., 1993, 1997; LeBaron et al., 1999).

Based on the conceptual model shown in Fig. 25, the tortuosity of the pathway should increase as the aspect ratio for the layered silicate increases. For example, Yano et al. (1993, 1997) proposed the following, relatively simple expression to describe the tortuosity factor,  $\tau$ , for gas-phase migration through a polymer-clay nanocomposite:

$$\tau = 1 + \left( \frac{L}{2W} \right) V_f \quad (9)$$

where  $L$  = the length of a dispersed monolayer of a layered silicate clay mineral,  $W$  = the thickness of the monolayer, and  $V_f$  = the volume fraction of the clay dispersed in the polymer. The  $L/W$  quotient in Eq. 9 represents the aforementioned aspect ratio for the layered silicate. With this expression for the tortuosity factor, the relative gas permeability for the nanocomposite can be written as follows (Yano et al., 1993, 1997):

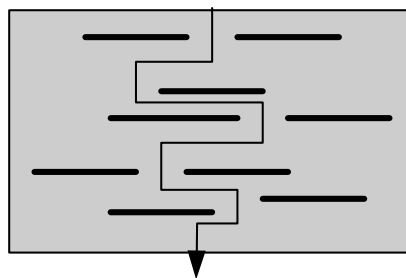


Figure 25. Schematic model for the tortuous pathway for diffusion of gas through an exfoliated polymer-clay nanocomposite (after Yano et al., 1993, 1997; LeBaron et al. 1999).

$$\frac{P_{nc}}{P_p} = \frac{1}{1 + \left(\frac{L}{2W}\right)V_f} \quad (10)$$

where  $P_{nc}$  and  $P_p$  = the permeability coefficients for the nanocomposite and the polymer matrix, respectively.

Based on this conceptual model, Yano et al. (1997) compared the measured and theoretical water vapor permeabilities of several polymer-clay nanocomposites consisting of four layered silicate clays (hectorite, saponite, montmorillonite, and synthetic mica) with different aspect ratios (i.e., values for  $L/W$ ) intercalated with an ammonium salt of dodecylamine and dispersed within a polyimide polymer. The results of their comparison in the form of the relative water vapor permeability,  $P_{nc}/P_p$ , are shown in Fig. 26.

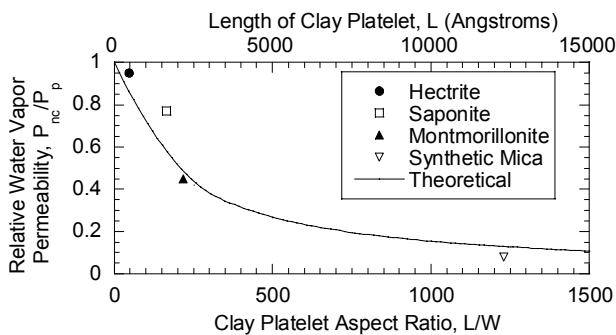


Figure 26. Measured and theoretical relative water vapor permeabilities of 60- $\mu$ m-thick films of polyimide-clay hybrid nanocomposites as a function of clay platelet aspect ratio (replotted after Yano et al., 1997; LeBaron et al., 1999).

Two conclusions can be drawn from the data shown in Fig. 26. First, the water vapor permeability of polymer-clay nanocomposites decreases as the aspect ratio of the layered silicate used in the nanocomposite increases. Second, the conceptual framework for the model represented by Fig. 25 and Eqs. 9 and 10 appears to provide theoretically determined values of  $P_{nc}/P_p$  that are reasonably close to the measured values of  $P_{nc}/P_p$ .

Due to significantly improved properties of polymer-clay nanocomposites relative to ordinary polymers, polymer-clay nanocomposites may offer a substantial improvement in barrier performance in environmental applications. For example, geomembranes comprised of polyethylene and polypropylene currently are widely used as barriers or components of barriers in waste containment systems, primarily due to their relative impermeability to liquids. However, VOCs are known to readily diffuse through geomembranes (Bonaparte et al., 2002; Edil, 2003). Thus, the use of polymer-clay nanocomposites comprised of a geomembrane with an organoclay dispersed within a polyethylene and polypropylene (i.e., geomembrane-clay nanocomposites) may offer substantial improvement in terms of the containment of VOCs. Also, because the molecules of VOCs typically are much larger than the molecules of common inorganic gases, the tortuosity effect and resulting reduction in the relative permeability of VOCs through geomembrane-clay nanocomposites may be even greater than that associated with the common inorganic gases as exemplified by the data in Fig. 24. Finally, although the relatively high cost of organoclays has restricted their use as barriers or barrier materials in practical applications, the amount of organoclay used in geomembrane-clay nanocomposites is sufficiently low (i.e., a few percent) such that the costs of the materials for geomembrane-clay nanocomposites should not be significantly greater than the cost of the geomembrane alone. However, the costs associated with the manufacture of geomembrane-clay nanocomposites will also have to be considered. Also, the use of geomembrane-clay

nanocomposites in waste containment applications remains innovative in the sense that no current data specifically addressing the use of geomembrane-clay nanocomposites exists.

## 5 EMERGING WASTE FORMS

The primary waste forms of concern over the past 25 years include inorganic chemical solutions containing toxic heavy metals (e.g., Cd, Pb, Zn), such as those derived from acid mine drainage (AMD), non-aqueous phase liquids that are either lighter than water (LNAPLs), such as the BTEX compounds (i.e., benzene, toluene, ethylbenzene, and xylene), or denser than water (DNAPLs), such as the chlorinated hydrocarbons (e.g., tetrachloroethylene, trichloroethylene), and radioactive wastes (Shackelford, 2002). In the vast majority of cases, the properties and characteristics required to evaluate the fate and mobility of these solutions and compounds are relatively well known, although their existence in mixtures (e.g., solid waste leachates) often complicates the picture. However, some waste forms have only recently been recognized, and undoubtedly additional waste forms that are currently unforeseen are likely to appear in the future. Thus, another potentially important issue facing the Geotechnical Engineering community is the issue of how to deal with emerging waste forms that recently have been recognized or will appear in the future. Two primary examples of such emerging waste forms include those derived from both living and dead animals, and those resulting from technological advances, such as the recent boom in the nanotechnology industry.

### 5.1 Animal wastes

Animal wastes can be classified into two categories, viz., wastes derived from living animals (e.g., manure and urine) as well as wastes derived from dead animals (e.g., carcasses). Both categories of waste are receiving increasing attention in terms of the potentially detrimental effects of such wastes on human health and the environment.

#### 5.1.1 Wastes from living animals

Over the last 20 years, animal agriculture has evolved to meet the needs of a rapidly growing population. One evolutionary trend in some countries such as the United States is the replacement of small to mid-size animal farms with large industrial-scale concentrated animal feeding operations (CAFOs). A diverse mixture of chemical and microbial substances is part of the animal waste stream at typical CAFOs. Much of the waste that is produced at these operations is ecologically benign organic matter that can be assimilated during the anaerobic digestion process. However, several classes of substances are of concern to industry and the regulatory community, including nutrients (e.g. nitrogen), pathogens (e.g. *E. coli* bacteria), and pharmaceuticals (e.g. antibiotics). The occurrence, fate, and transport of these substances are similarly of significant interest to industry, research and regulatory communities.

As a result of the need to protect the environment from these substances, animal wastes from CAFOs involving cattle, swine, and poultry are usually stored in anaerobic lagoons or storage basins that typically are lined with up to 0.91 m of a low permeability compacted clay liner (CCL). In the context of these animal waste containment systems, nutrients such as nitrogen and pathogens have been studied the most (e.g., Reddi and Davalos, 2000), primarily because both of these contaminants typically are in relatively high concentrations in animal wastes. However, because of the increasingly intensive use of antibiotics in the management of CAFOs, primarily for growth promotion, the potential exists for the transport of these compounds and their metabolites into surface and ground waters.

For example, one half of the 22.7 million kilograms of antibiotics produced each year in the United States is used for agriculture, and 90 % of these antibiotics are used for growth promotion. Some of the drugs are completely metabolized to inactive compounds, but many are excreted as active metabolites. As a result, several studies have recently been undertaken to quantify the occurrence of these compounds in animal waste and the subsequent transport to surface and ground waters (Kolpin et al., 1999; Meyer et al., 1999; Thurman and Hostetler, 1999; Meyer et al., 2000).

For example, several classes of antibiotics (e.g., tetracycline, sulfonamides, beta-lactams and macrolides) have been found to be present in hog waste lagoons at concentrations as high as 0.7 mg/L (Meyer et al., 1999). Antibiotics also have been detected in wells near hog waste lagoons indicating that the compounds are being transported across the clay liners and through the subsurface (Meyer et al., 1999). Finally, Meyer et al. (2000) used a commercially available radioimmunoassay to analyze liquid waste samples obtained from 13 hog-waste lagoons in three states. In addition, samples were analyzed from 52 surface and groundwater areas associated with intensive swine and poultry production in seven states. The tests yielded positive results for tetracycline antibiotics in samples from all 13 of the hog lagoons with results ranging from approximately 5 to several hundred ppb in the liquid waste. All but two of the 52 surface and groundwater samples tested negative, and these two samples contained tetracycline antibiotic concentrations less than 1 ppb. The presence of antibiotics in the animal waste stream was confirmed, but the transport of these antibiotics through animal waste containment liners, such as CCLs and GCLs, has not been studied. Thus, the issue of the mobility and fate of these antibiotics during migration through containment systems, and the resulting effectiveness of such containment systems remains an environmental issue that geotechnical engineers will have to address.

#### 5.1.2 Wastes from dead animals

The relatively recent outbreak of a wide variety of animal transmitted diseases, such as Avian Influenza (Bird Flu), Mad Cow Disease (MCD), Chronic Wasting Disease (CWD), Newcastle Disease (ND) and Severe Acute Respiratory Syndrome (SARS), has raised increasing concern regarding the potential transmission of these diseases to humans and the resulting potentially harmful effects. Both MCD (or bovine spongiform encephalopathy, BSE) and CWD (which affects deer and elk) belong to a group of diseases known as transmissible spongiform encephalopathies (TSEs) ([www.mad-cow.org/](http://www.mad-cow.org/), [www.cdc.gov/flu/avian/](http://www.cdc.gov/flu/avian/), [www-micro.msb.le.ac.uk/3035/prions.html](http://www-micro.msb.le.ac.uk/3035/prions.html), [www.aphis.usda.gov/vs/nahps/cwd/](http://www.aphis.usda.gov/vs/nahps/cwd/)). Other TSEs include scrapie of domestic sheep, Creutzfeldt-Jakob disease (CJD) of humans, and a new variant of CJD (called new variant Creutzfeldt-Jakob diseases, or nvCJD) that is probably linked to the BSE agent. The TSE diseases typically are characterized as brain diseases where small holes develop in the brains of the animals that eventually develop into an appearance resembling a sponge with large holes. The process results in loss of motor control, dementia, paralysis wasting and eventually death.

Besides CJD, humans are also susceptible to several other TSEs, including GSS (Gerstmann-Straussler-Scheinker syndrome), FFI (Fatal familial Insomnia), Kuru, and Alpers Syndrome, and dozens of people in England are thought to have contracted the human form of BSE (i.e., nvCJD) by eating the meat of infected cattle. Although the incidences of TSEs in humans currently are thought to be low, the relatively long time-to-death associated with the diseases, which are on the order of 10 or more years from the time of infection, and the fact that there currently is no known cure for TSEs, which are always fatal to susceptible host species, cause concern for the future.

The primary culprit causing TSEs is thought to be prions

(pronounced *pree-ons*), which can be defined as small proteinaceous infectious particles that resist inactivation by procedures that modify nucleic acids ([www.microbe.org/news/prions.asp](http://www.microbe.org/news/prions.asp)). Examples of prions from hamsters and mice are shown in Fig. 27. Some prion diseases are of particular concern because they are not only infectious but also hereditary. Thus, the infection can be passed on from generation to generation. In addition, prions are particularly recalcitrant and persistent, and can survive after being steamed, frozen, disinfected, zapped with ultraviolet light or bombarded with X-rays (Grady, 2004). Thus, disposed carcasses of diseased animals in landfills represent a source of prions that can eventually end up in surface and ground waters if not contained properly. Consequently, prions represent an emerging waste form that must be characterized with respect to fate and mobility in order to ensure that waste containment systems are effective in protecting human health and the environment from contracting TSEs.

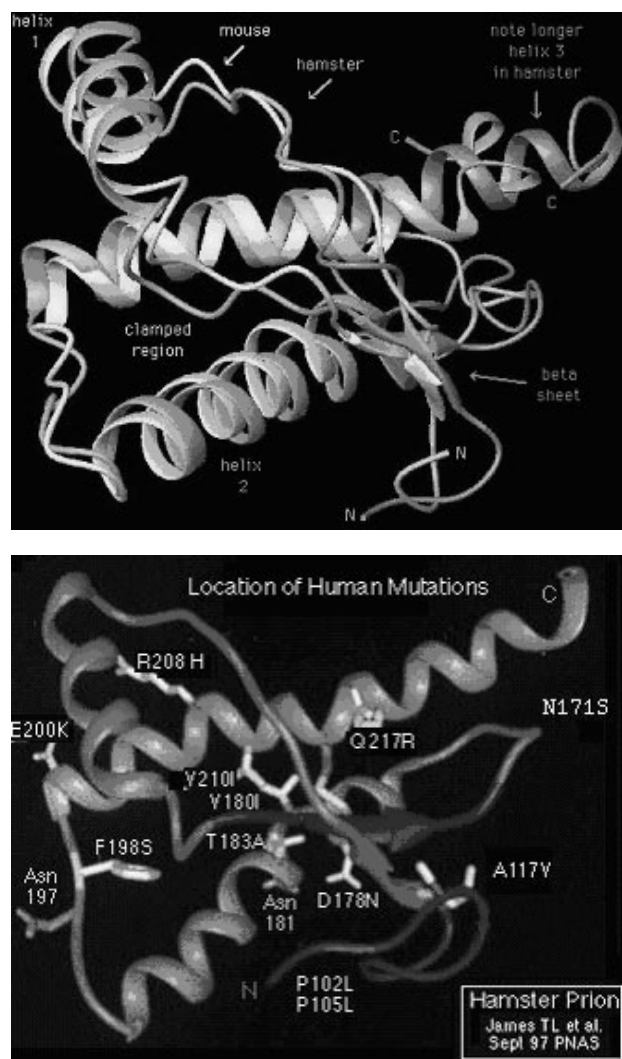


Figure 27. Examples of hamster and mouse prions (from: The Official Mad Cow Disease Home Page; [www.cyberdyne.com/%7Eetom/mad\\_cow\\_disease.html](http://www.cyberdyne.com/%7Eetom/mad_cow_disease.html)).

#### 5.2 New technology derived wastes

A primary example of this category of emerging waste is the relatively recent interest in nanotechnology, or technology dealing with scales on the order of 1 to a 100 nanometers (1 nm =  $10^{-9}$  m). An explosion of inventions and breakthroughs has recently occurred in nanotechnology in a number of industries, including the chemical, medical, electronic, computer and food industries (Dionysiou, 2004). However, the advent of nanotech-

nology also has raised issues about the potential for negative environmental and health implications resulting from the concomitant production of "nanowaste". Such issues include (Dionysiou, 2004):

- (1) the toxicity of manufactured nanoparticles or nanofibers for humans, animals, fish, and other ecological impacts, such as plants and crops;
- (2) the production, use, and fate of nanomaterials through life-cycle analysis;
- (3) the role of benign nanomaterials released into the environment in terms of facilitating the transport of pollutants;
- (4) the biodegradability and persistence of nanomaterials;
- (5) the potential sources for releasing toxic nanomaterials into the environment;
- (6) the effectiveness of existing methods for treating and/or removing toxic nanomaterials from the environment; and
- (7) the intentional use of nanomaterials as weapons in military and terrorist operations.

Of particular interest to geotechnical engineers is the issue of whether the current technology for waste containment systems is sufficient to deal with any new technology derived waste forms and, if not, what additional waste containment technologies will be required.

An example of the fate and transport of nanomaterials is given by the results of the study by Lecoanet et al. (2004), who evaluated the mobility of a wide range of nanomaterials, including two sizes of silica, anatase, ferroxane, alumoxane, fullerol (hydroxylated  $C_{60}$ ), clusters of  $C_{60}$  referred to as n- $C_{60}$ , and single-wall carbon nanotubes (SWNT), during migration through a short column ( $L = 9.25$  cm) of silicate beads at a flow rate of 12 mL/min. The results of their experiments are reproduced in Fig. 28 in the form of nanoparticle breakthrough curves (BTCs), or relative concentration in the effluent versus pore volumes of flow (PVF). An interesting observation from the data shown in Fig. 28 is that several of the nanoparticles (e.g. alumnox, silica-135, fullerene, anatase, and ferrox) exhibit trends that resemble those for chemical solute species that undergo first-order decay. However, the nanoparticles are not subject to decay, but rather are deposited and 'attached' to the surfaces of the particles comprising the immobile porous media, referred to as the 'collector', via an attachment efficiency factor,  $\alpha$ , that can be defined as follows (Lecoanet et al., 2004):

$$\alpha = -\frac{2d_c}{3(1-\varepsilon)\eta_0 L} \ln\left(\frac{c}{c_0}\right) \quad (11)$$

where  $d_c$  = the diameter of the collector (assumed to be spherical) in the porous medium,  $\varepsilon$  = the porosity of the porous medium,  $L$  = the length of the porous medium,  $c$  = the nanoparticle concentration at  $L$ ,  $c_0$  = the source nanoparticle concentration (i.e., at  $x = 0$ ), and  $\eta_0$  = the clean bed single collector efficiency that describes the particle transport to an individual collector before particle accumulation alters the geometry. The expression given by Eq. 11 is similar to that for first-order decay (e.g., Shackelford and Rowe, 1998), or

$$\ln\left(\frac{c}{c_0}\right) = -\lambda t \quad (12)$$

where  $t$  = time, and  $\lambda$  = the decay constant, which typically is related to the half-life for decay,  $t_{0.5}$ , as follows:

$$\lambda = \frac{0.693}{t_{0.5}} \quad (13)$$

By equating Eqs. 11 and 12, we can see that

$$\lambda = \frac{3\alpha(1-\varepsilon)\eta_0 L}{2d_c t} \quad (14)$$

Thus, the possibility of describing the fate of nanoparticles during migration through porous materials based on first-order decay exists, although the underlying principles for this correlation are not well understood at present. The need to understand such principles for geotechnical applications such as waste containment is an environmental issue yet to be resolved.

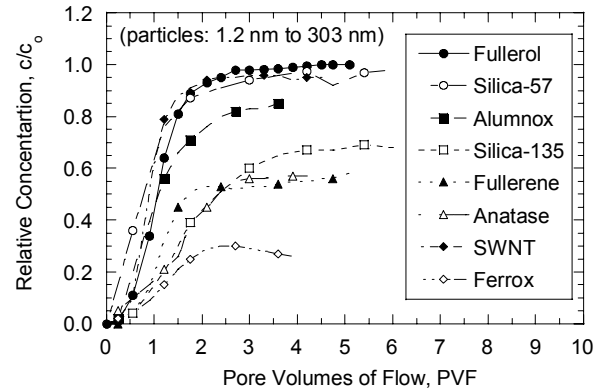


Figure 28. Breakthrough curves of nanoparticle suspensions migrating through a short column of silicate beads (data from Lecoanet et al., 2004).

## 6 SIGNIFICANCE OF BIOLOGICAL PROCESSES

Another significant environmental issue of concern to geotechnical engineers is the recognition of the increasing importance of biological processes in governing the performance of geotechnical engineered systems, as described in a recent and comprehensive review by Mitchell and Santamarina (2005). As noted by Mitchell and Santamarina (2005), biological processes, in general, and microbiological processes, in particular, play a significant role in dictating soil behavior, although the effect of biological activity on soil mechanical behavior remains largely under explored in Geotechnical Engineering. Aside from the obvious role of biological processes in bioremediation, biological processes also are important in the settlement of municipal solid waste (MSW) in landfills (Park and Lee, 1997; El-Fady and Khoury, 2000; Park et al., 2002; Marques et al. 2003), the clogging of porous media, such as leachate collection and removal systems (Fleming et al., 1999; Rowe et al., 2000a,b; Cooke et al., 2001; VanGulck and Rowe, 2004; Tumuluri et al., 2005), the reduction in hydraulic conductivity of compacted clay liners (Kamon et al., 2002), and the management and operation of bioreactor landfills (Reinhart et al., 2002; Benson et al., 2004). In terms of bioremediation, recent emphasis also has been placed on stimulating native bacteria (biostimulation) in low permeability soils contaminated with organic compounds that are subject to biodegradation by driving nutrients into the soils via electrokinetics (EK) under DC electrical fields (Alshawabkeh and Maillacheruvu, 2001; Wu et al., 2003), as well as the use of sulfate reducing bacteria (SRB) in permeable reactive walls (PRWs) for *in situ* treatment of acid mine drainage (U.S. EPA, 1998; Benner et al., 1999, 2000; Blowes et al., 2000; Gobla, 2002; Ludwig et al., 2002). A brief description of some of these applications is provided to illustrate the importance of biological processes in environmental applications of Geotechnical Engineering and some of the issues arising from this importance.

## 6.1 Bioreactor landfills

Bioreactor landfills may be defined as landfills that are operated to enhance the degree and rate of waste decomposition by addition of water and recirculation of leachate. The addition of water and recirculated leachate stimulates microbial activity by providing better contact between insoluble substrates, soluble nutrients, and microorganisms (Barlaz et al., 1990). Bioreactor landfills have several potential advantages relative to conventional landfills (Benson et al., 2004):

- (1) the long-term risk resulting from the persistence of leachate and gas generation in conventional landfills is reduced;
- (2) the enhanced decomposition in bioreactor landfills increases the rate of MSW settlement thereby providing for additional storage volume while reducing the total land use for landfills, as well as the potential for long-term settlement-induced damage of the final cover;
- (3) the viability of gas-to-energy options is improved due to the increase in the rate of landfill gas production; and
- (4) the reduction in leachate treatment costs is possible with recirculation of leachate.

With respect to (1), final covers in traditional landfills are used, by definition, to minimize the moisture ingress into the waste that is essential to biodegradation. Consequently, waste in a traditional landfill is contained or entombed and remains practically intact for long periods of time, possibly in excess of the design life including the post-closure monitoring period of the containment system (Reinhart et al., 2002). Thus, accelerating the process of waste biodegradation by recirculating water, leachate, and/or other amendments (e.g., biosolids, nutrients) provides for more rapid stabilization of the waste and a reduction in the long-term risk associated with prolonged production of leachate and gas.

The essential elements of waste containment systems required for operation of bioreactor landfills include (Reinhart et al., 2002): (1) a leachate collection system, (2) a liner, (3) a gas collection system, and (4) controlled moisture introduction. The last element, viz. controlled moisture introduction, has been recognized as the single most important factor in enhancing waste decomposition in landfills (Pohland, 1980; Reinhart et al., 2002). In this regard, leachate recirculation has been found to be the most practical approach for controlling the moisture content of the waste. The common methods for recirculating leachate, as shown schematically in Fig. 29, include (Reddy and Bogner, 2005): (a) spraying, (b) surface ponding and infiltration, (c) horizontal wells in trenches, and (d) vertical wells. The type of recirculation system used and the corresponding method of operation are a function of the desired moisture distribution, the ability to minimize environmental impact, and regulatory compliance (Reinhart et al., 2002).

Although the concept of recirculating leachate to enhance the rate of waste biodegradation has been around for approximately 30 years, and the number of landfills recirculating leachate in the United States is increasing, the percentage of bioreactor landfills relative to traditional landfills has remained relatively constant at about 5 to 10 % (Reinhart et al., 2002). The reluctance to use the bioreactor landfill technology can be traced to several factors, including a perception that the technology is not well demonstrated, technical impediments, unclear cost implications, and regulatory constraints. For example, although the landfill regulations in the United States under RCRA Subtitle D have long permitted leachate recirculation at lined landfills, the possibility of injecting liquids not generated in the landfills has only recently (March 2004) been allowed through regulation referred to as the Research, Development and Demonstration [RD&D] permits to landfills (Tom, 2004). Also, some of the technical impediments have included uncertainties pertaining to

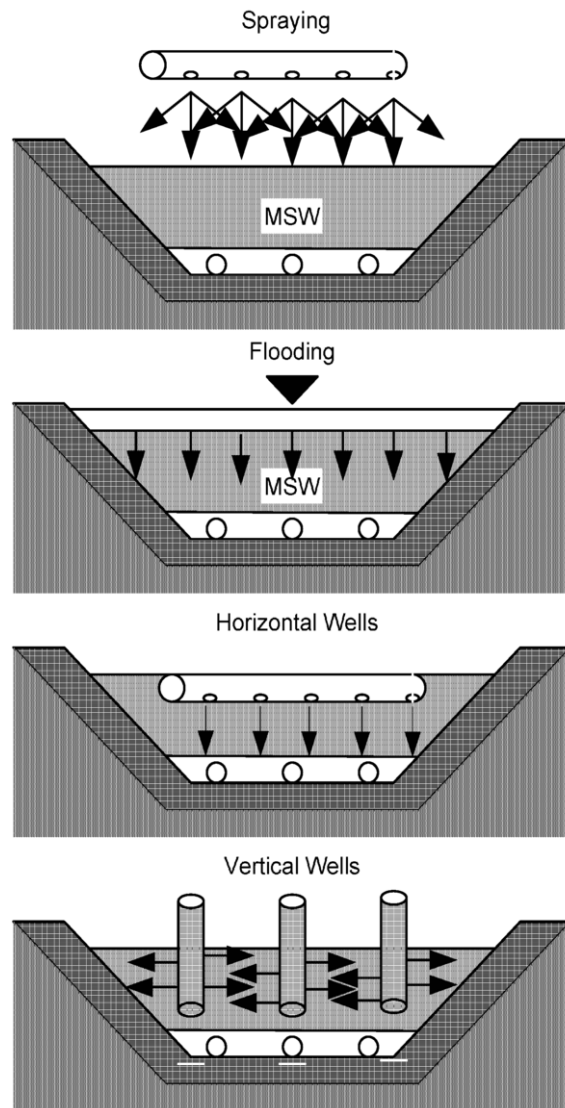


Figure 29. Approaches for applying re-circulated leachate in bioreactor landfills.

the capture of landfill gas, the treatment and storage of leachate, reuse of landfill space and capacity, abatement of greenhouse gas, bioreactor design, consideration of solid waste density, determination of leachate/moisture distribution within the waste, degradation, pretreatment and management of the waste, waste compressibility and settlement, daily cover considerations, and management of amendments (Reinhart et al., 2002; Reddy and Bogner, 2005).

Nonetheless, interest in the bioreactor landfill technology has increased substantially as is evident by the vast number of recent studies aimed at improving our understanding and feasibility of bioreactor landfill technology (e.g., O'Keefe and Chnowethg, 2000; Bogner et al., 2001; San and Onay, 2001; Yuen et al., 2001; Mehta et al., 2002; Warith, 2002; Gachet et al., 2003; Hupe et al., 2003; Price et al., 2003; Al-Thani et al., 2004; Sponza and Agdag, 2004; Bureau et al., 2005; Chenu et al., 2005; Daniels et al., 2005; de Abreu et al., 2005; Gabr et al., 2005; Hansen, 2005; Jain et al., 2005; Khire and Haydar, 2005; Reddy and Bogner, 2005). Some of the important geotechnical issues that currently are being addressed include evaluating the changes in waste compressibility as a function of the state of decomposition (Hossain et al., 2003), evaluating the use of a geocomposite drainage layer to recirculate or inject leachate (Khire and Haydar, 2005), and determining the relative

changes in waste shear strength during waste decomposition for the purpose of evaluating stability of the waste (Gabr et al., 2005). These and other issues will require further evaluation and resolution before wide-spread acceptance of the bioreactor land-fill technology is forthcoming.

## 6.2 Bioremediation with electrokinetics

Electrokinetic remediation refers to the process whereby a low-voltage, direct-current (DC) electrical field is applied across a section of contaminated soil to move contaminants or modify soil conditions, such as pH, Eh and DO, for bioremediation. The electrical field is applied through electrodes (anode and cathode) that are inserted into the soil, and charged particles, such as ionic chemical species, are mobilized by the applied electrical current. The use of electrokinetics (EK) as an effective remediation technology for removing contaminants from relatively low permeability ( $< 10^{-5}$  cm/s), fine-grained porous media is relatively well established (e.g., Acar and Alshawabkeh, 1993, 1996; Mitchell, 1993), and numerous studies pertaining to the technology have been performed in the last 10-15 years. Most of the earlier studies focused primarily on the removal of ionic contaminants, such as heavy metals (e.g., Hamed et al., 1991; Pamukcu and Wittle, 1992; Probst and Hicks, 1993; Runnells and Wahli, 1993; and Pamukcu et al., 1997). However, more recently, the interest in EK has focused on the injection and transport by electrical fields of additives that can enhance bioremediation of contaminated fine-grained soils under aerobic or anaerobic conditions (e.g., Acar et al., 1996; Alshawabkeh and Maillacheruvu, 2001; Sarahney and Alshawabkeh, 2005). This concept is illustrated schematically in Fig. 30.

As shown in Fig. 30, application of a DC field to electrodes inserted into the clay results in the migration of positively charged ions (cations) towards the negatively charged electrode (cathode), and the migration of negatively charged ions (anions) towards the positively charged electrode (anode). This transport process is referred to as ion migration. Simultaneously, displacement of the cations that dominate the diffuse double layer of the clay particles under the imposed electrical field towards the cathode results in bulk pore water movement via a process known as electro-osmosis. The two processes of ion migration and electro-osmosis are the primary processes governing chemical transport under electrical fields in consolidated clay deposits (Alshawabkeh and Acar, 1992, 1996; Acar and Alshawabkeh, 1993). However, as illustrated schematically in Fig. 30, microorganisms (typically negatively charged) also can be driven into the soil via electrophoresis to enhance bioremediation of organic compounds (e.g., DeFlaun and Condee, 1997), such as chlorinated hydrocarbons (TCE) and the BTEX compounds, a process often referred to as bioaugmentation. Alternatively, bioremediation can be enhanced via biostimulation whereby nutrients, such as nitrogen and phosphate, are injected and transported into the soil under electrical fields to stimulate the activity of microbial cultures already present in the soil. For example, recent studies have evaluated the potential use of organic acids, such as lactic acid ( $C_3H_6O_3$ ), as stimulants via EK injection and transport for the bioreductive dechlorination of chlorinated hydrocarbons, such as tetrachloroethylene, carbon tetrachloride, and trichloroethane (e.g., Wu et al., 2003, 2005).

Because the use of the EK technology for *in situ* bioremediation is a relatively new concept, several issues need to be addressed before wide-spread acceptance of the technology is gained and full-scale implementation of the technology can proceed (see Alshawabkeh and Maillacheruvu, 2001). For example, application of the electrical current generates electrolysis reactions at both electrodes, resulting in acid conditions (low pH) and generation of oxygen at the anode via oxidation, and

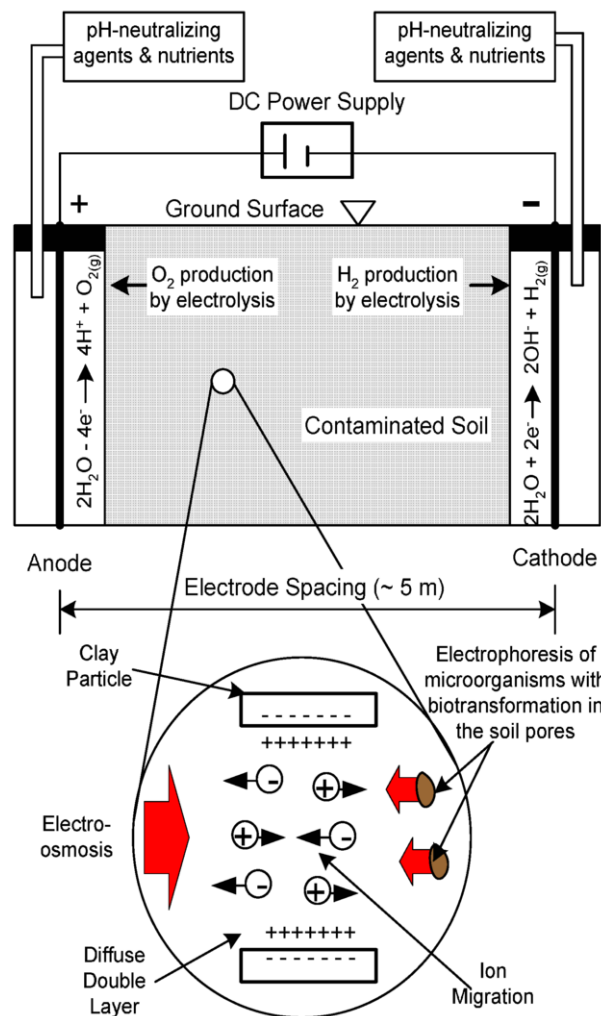


Figure 30. Schematic of the electrokinetic system for bioremediation (redrawn from Alshawabkeh and Maillacheruvu, 2001).

basic conditions (high pH) and generation of hydrogen gas at the cathode via reduction (see Fig. 30). The generation of oxygen at the anode can help stimulate aerobic biodegradation, but may adversely impact anaerobic biodegradation. In addition, the production of protons ( $H^+$ ) at the anode can drop the pH to below 2 resulting in conditions that are unfavorable for, and even harmful to, microbial growth. In this case, the use of pH neutralizing agents at the electrodes, such as ammonium hydroxide at the anode and organic acids at the cathode, may be required (Acar et al., 1996; Alshawabkeh and Maillacheruvu, 2001). Other significant issues related to the use of EK for bioremediation pertain to the impacts of the DC electrical fields on (1) the biogeochemical interactions within the soil prior to application of the technique, (2) microbial adhesion and transport in the subsurface, and (3) the activity of the microorganisms in the soil matrix (Alshawabkeh and Maillacheruvu, 2001).

With respect to issue (3), Alshawabkeh and Maillacheruvu (2001) describe the results of a study indicating that anaerobic cultures appeared to experience "environmental shock" when exposed to electrical field intensities  $> 1.4$  V/cm. However, these cultures were able to recover their activity after the electrical currents were turned off. On the contrary, aerobic cultures were not adversely affected when exposed to electrical field intensities  $< 0.28$  V/cm. However, exposure of aerobic cultures to DC field intensities up to 1.14 V/cm initially stimulated activity up to a period of 24 hrs, whereas exposures exceeding 24 hrs appeared to retard growth (Alshawabkeh et al., 2004). These re-

sults illustrate the complicated interactions that can occur when biological processes are considered, and the potential significant of these interactions in terms of bioremediation with electrokinetics.

### 6.3 Remediation with sulfate reducing bacteria

A number of laboratory experiments have demonstrated the removal of metals from solution due to sulfate reduction via sulfate reducing bacteria (SRB) and metal-sulfide precipitation. These experiments usually employed solid, decomposable organic materials as sources of nutrients for SRB that reduce sulfate ( $\text{SO}_4^{2-}$ ) to sulfide ( $\text{S}^{2-}$ ) which, in turn, reacts with toxic heavy metals to form metal sulfides that precipitate from solution. Examples of the solid, decomposable organic materials that have been tested include sawdust (e.g., Tuttle et al., 1969; Wakao et al., 1979), spent mushroom compost (e.g., Dvorak et al., 1992; Hammack and Edenborn, 1992), fresh alfalfa (e.g., Bechard et al., 1994), whey/cow manure (e.g., Christensen et al., 1996), leaf compost/sewage sludge (e.g., Prasad et al., 1999), spent mushroom compost/oak chips/wastepaper sludge (e.g., Chang et al., 2000), leaf compost/sawdust/wood chips/sewage sludge (e.g., Waybrant et al., 1998; Waybrant et al., 2002), leaf compost/wood chips/poultry manure (e.g., Cocos et al., 2002), wood chips/pulp and paper waste (Hulshof et al., 2003), and wheat straw (Frommichen et al., 2003), among others. In some cases, organic solid materials have been amended with dissolved organic substrates (e.g., Bechard et al., 1994; Frommichen et al., 2003).

Based largely on the successes of these laboratory studies, several field applications employing the use of SRB in permeable reactive trenches (PRTs) and bioreactors based on solid decomposable organic materials for remediation of acid mine drainage (AMD) have been reported (e.g., U.S. EPA, 1998; Benner et al., 1999, 2000, 2002; Blowes et al., 2000; Gobla 2002; Ludwig et al., 2002). For example, a sulfate reducing PRT containing 50 % gravel, 20 % municipal compost, 20 % leaf mulch, 9 % wood chips, and 1 % limestone, all by volume, was constructed in 1995 at the Nickel Rim mine in Ontario, Canada (Benner et al., 1999, 2000; Blowes et al., 2000). The primary advantages to this approach for remediating AMD are that (1) the organic substrate materials are relatively inexpensive, (2) the process of sulfate reduction is naturally occurring in the presence of SRB, and (3) the operational costs associated with passive treatment technologies, such as engineered wetland systems and PRTs typically are substantially lower than those associated with more active, intrusive remediation technologies.

However, there are several issues related to using SRB for remediating AMD. Probably the most significant issue relates to the ability of the treatment system to sustain the biological activity over the typically prolonged durations required to treat large contaminant plumes with passive systems like PRTs. This ability to sustain the biological activity is limited in part by the availability of carbon used by the SRB via decomposition of the solid organic substance. A limited rate and/or amount of carbon can result in eventual shut down of the reactive treatment system. Another issue pertains to the need to buffer the pH of AMD, which is typically quite low ( $\text{pH} < 2$ ), to a more neutral value (typically  $6 < \text{pH} < 8$ ) that is more favorable for biological activity.

A third issue with this approach is simply the complexity of the treatment system. For example, as shown in Fig. 31, the primary processes associated with sulfate reduction and metal precipitation using SRB include (Hemsi et al., 2005): (i) anaerobic decomposition of cellulose/hemicellulose in solid particulate organic materials due to the activity of decomposer bacteria, producing lactate, with hydrolysis as the rate limiting step; (ii)  $\text{SO}_4^{2-}$  reduction based on the incomplete oxidation of lactate (sulfate reduction 1 in Fig. 31); (iii) instantaneous or kinetically controlled precipitation of metal ( $\text{Me}^{2+}$ ) sulfides; and (iv) partial volatilization of  $\text{H}_2\text{S}$  to the gas phase. Additional processes in-

clude: (i)  $\text{SO}_4^{2-}$  reductions based on the complete oxidation of lactate and acetate (sulfate reductions 2 and 3), and (ii) competing processes of lactate oxidation and methanogenesis. As indicated, most of these processes are kinetically controlled. Due to the complexity of the overall sulfate reduction processes with SRB, and the myriad of possible solid organic material compositions that can be used in these treatment systems, a generalized systematic approach at design and operation of these systems remains an issue.

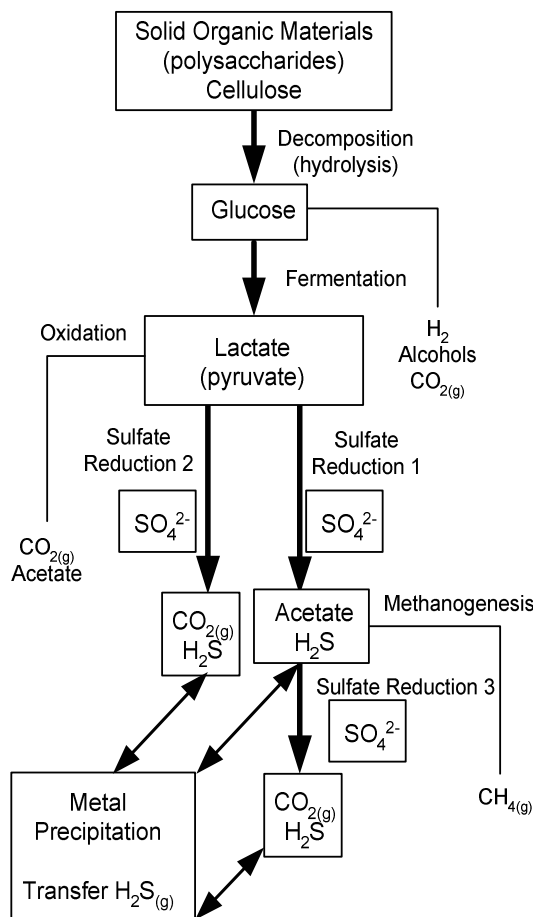


Figure 31. Possible biochemical reactions for metals precipitation resulting from sulfate reduction in aqueous solution (redrawn from Hemsi et al., 2005).

## 7 ROLE OF MODELING

The ability of models to accurately predict field performance of engineered systems has been and will continue to be an issue facing geotechnical engineers. This issue is particularly important when dealing with environmental problems because models often are used as a tool to predict the future impacts and resulting risk from engineering activities related to environmental protection, such as new waste disposal facilities and existing sources of contamination (e.g., waste dumps and piles, accidental chemical spills, etc.). The results of such predictions may be required for regulatory acceptance of a chosen approach and, as a result, are often highly scrutinized due to the potential for detrimental impacts on human health and the environment. The uncertainty in the accuracy of the model predictions can be considerable, particularly in cases where the time-frame for the predictions is long, such as the disposal of radioactive wastes with design lives ranging from hundreds to thousands of years. In such cases, uncertainty is associated not only with the comprehensiveness and accuracy of the physical, chemical, and bio-

logical processes upon which the model is based, but also with the accuracy of the input data and the lack of knowledge concerning the changes in the material properties and processes that may occur with time. As a result, predictions made with existing models generally cannot be considered reliable until and unless the predictions are verified by comparison with field data.

In this regard, and as an example, a joint research effort between the University of Wisconsin-Madison and Colorado State University is currently being conducted to evaluate the use of existing unsaturated flow models to predict the hydrologic performance of alternative covers (e.g., see Benson et al., 2004, 2005). Although the underlying principle of alternative covers is simple, accurately predicting the hydrologic performance of alternative covers is difficult, at best, because flow of water in unsaturated soils, and therefore alternative covers, is a highly non-linear process that is subjected to boundary conditions that reflect the natural randomness of precipitation events. In addition, water removal via plant transpiration and evaporation to the atmosphere is not fully understood, particularly in semi-arid and arid locations. Sophisticated numerical models simulating these processes have been developed for alternative cover design, but only limited effort has been devoted towards ensuring that predictions made with hydrologic models accurately represent field conditions (Fayer et al., 1992; Fayer and Gee, 1997; Khire et al., 1997, 1999; Roesler et al., 2002; Scanlon et al., 2002; Benson et al., 2004, 2005).

As a result of these considerations, five models commonly used for the design of alternative covers are being evaluated in the study (Wilson et al., 1999). The models being considered are as follows:

- **HELP (Hydrologic Evaluation of Landfill Performance;** Schroeder et al., 1994);
- **UNSAT-H (Unsaturated Water and Heat Flow;** Fayer 2000);
- **VADOSE/W (Newman, 2002);**
- **HYDRUS-2D (Simunek et al., 1996); and**
- **LEACHM (Leaching Estimation and Chemistry Model;** Hutson and Wagenet, 1996).

These five models can be divided into two different categories: *water balance models* (HELP) and *Richards' equation models* (UNSAT-H, HYDRUS-2D, LEACHM, VADOSE/W). Water balance models partition precipitation into runoff and infiltration using empirical models, and route water through the barrier using simplified algorithms. In contrast, Richards' equation models employ Richards' partial differential equation for unsaturated flow to describe how water enters and moves through the cover, and to predict the rate at which water percolates into the underlying waste (Khire et al., 1997, 2000; Wilson et al., 1999). The primary advantages of the water balance models are simplicity and ease of use, whereas the primary advantage of the models based on Richards' equation is believed to be better accuracy, as these models are based on the fundamentals of the physics of water migration under unsaturated conditions (Fayer and Gee, 1997; Khire et al., 1997, 2000).

The results of simulations using each of the aforementioned models are being compared with the field measured data for alternative covers at a selected number of the sites in the aforementioned ACAP (see Fig. 17). Measured hydraulic properties from Gurdal et al. (2003) are being used as input along with field-measured properties of the vegetation. On-site data are used for the meteorological input. Hydraulic parameters for each soil type have been defined for four cases: (1) mean storage capacity (MSC), (2) high storage capacity (HSC), (3) low storage capacity (LSC), and (4) field fit (FF). The MSC case was defined using the geometric mean of log-normally distributed parameters (saturated hydraulic conductivity, van Genuchten's  $n$  parameter) and the arithmetic mean of normally distributed parameters (saturated and residual water contents, van

Genuchten's  $n$  parameter). Parameters corresponding to "high" and "low" storage capacity were defined as two standard deviations from the mean (geometric or arithmetic, depending on the parameter). High storage capacity was assumed to correspond to high  $n$  and saturated water content along with low  $\alpha$ , saturated hydraulic conductivity, and residual water content. Similarly, low storage capacity was defined using low  $n$  and saturated water content along with high  $\alpha$ , saturated hydraulic conductivity, and residual water content. Field-fit parameters were derived from fitting hydraulic functions to water content and matric suction measured in the field.

Examples of the comparisons among the water balance predictions using the four models based on Richards' equation (i.e., UNSAT-H, HYDRUS-2D, LEACHM, VADOSE/W) for the MSC case at the Altamont, CA, and Boardman, OR, sites (see Figs. 15 and 17) are shown in Figs. 32 and 33, respectively. A quick glance at the comparisons in Figs. 32 and 33 reveals that, although the models tend to mimic the trends in the measured data fairly well, the accuracy of the predictions ranges from poor to good based on the model and the water balance parameter being considered.

For the Altamont site (Fig. 32), VADOSE/W and HYDRUS-2D provide relatively accurate fits to surface runoff (SRO), whereas UNSAT-H substantially over estimates SRO and LEACHM significantly under estimates SRO. One reason for the substantial over estimation in SRO based on UNSAT-H is that the simulation was performed using the default 1 cm/hr precipitation rate, which is a much higher rate than was used with the other models. An under estimation in SRO, such as that indicated by the results using LEACHM, is a relatively uncommon occurrence and can be attributed, in part, to a higher hydraulic conductivity at air entry suction for the surface layer assumed in LEACHM due to the difference in the hydraulic conductivity function implemented in LEACHM (i.e., Campbell) relative to all the other models (van Genuchten-Mualem). An inaccurate prediction in SRO implies that infiltration is incorrectly predicted (e.g., infiltration is the complement to surface runoff for UNSAT-H and VADOSE/W, since both models ignore interception by the plant canopy) such that the volume of water to be managed by the cover is incorrect, and all subsequent water balance quantities will be incorrect (Benson et al., 2004, 2005).

In terms of soil-water storage (SWS) at Altamont, during periods of maximum SWS (i.e., during the winters), the SWS predicted using both VADOSE/W and HYDRUS-2D tends to mimic reasonably well the trends in SWS, whereas UNSAT-H under predicts SWS and LEACHM over predicts SWS, probably due, in part, to the over prediction in SRO by UNSAT-H and the under prediction in SRO by LEACHM. However, during periods of minimum SWS (i.e., during the summers), all four models tend to under predict SWS, with the under prediction being greatest during the Spring and Summer of 2003.

The significant difference between the field and predicted SWS by all four models during the Spring and Summer of 2003 is attributed to greater than typical precipitation that occurred during the Winter of 2003 which exceeded the transpiration capacity of the plants allowing the water to accumulate (Benson et al., 2004, 2005). The phenology of the vegetation also appears to have been altered, with transpiration ceasing much earlier than normal (hence, the lack of a drop in measured SWS during spring and summer). Changes in phenology in response to changes in climate are not taken into account by most cover models (including the four models considered herein). As a result, model predictions exhibit a similar pattern of water extraction each year regardless of the current climate, which can result in gross deviations in SWS such as those shown at the end of the record in Fig. 32 (Benson et al., 2004, 2005).

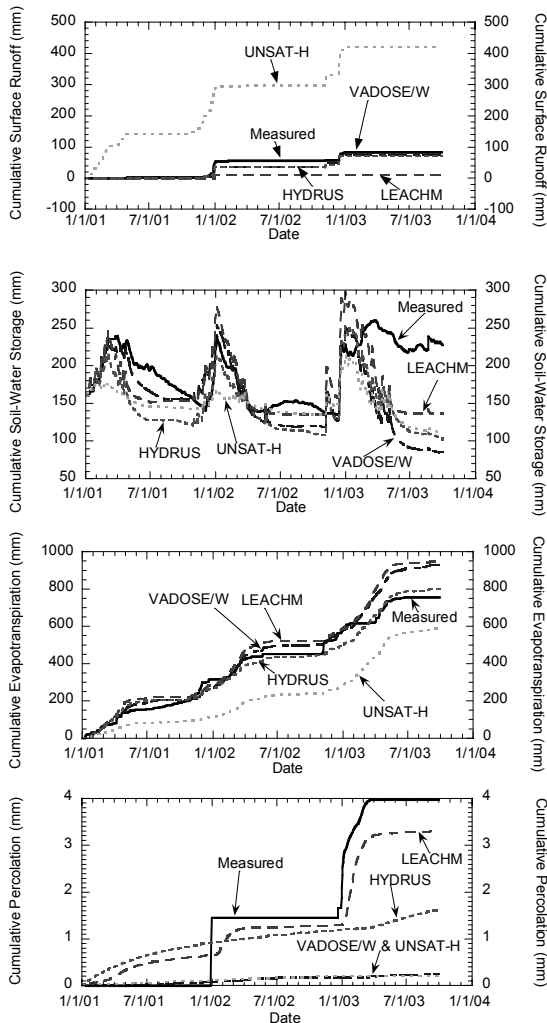


Figure 32. Measured versus predicted water balance parameters based on four numerical unsaturated flow models for the monolithic cover constructed at Altamont, CA.

In terms of evapotranspiration (ET) at Altamont, UNSAT-H severely under predicts ET, which is a direct consequence of the over prediction in SRO by UNSAT-H; i.e., too little water has entered the soil and is available for evaporation or transpiration (Benson et al., 2004, 2005). However, the other three models provide reasonably accurate predictions in ET, with HYDRUS-2D providing the most accurate predictions of ET and LEACHM and VADOSE/W slightly over predicting ET. The slight over estimation in ET by LEACHM can be attributed primarily to the under estimation in SRO by LEACHM, whereas the slight over estimation in ET using VADOSE may be attributed to several possible factors, including the fact that the algorithm for transpiration not capturing the apparent "shut down" of the plants in the Spring and Summer of 2003.

Comparison of the measured and predicted percolation for Altamont reveals that only LEACHM provides relatively accurate fits to the measured percolation both in terms of the trends in the data and the magnitude of the data (i.e., considering the narrow range in data). Both VADOSE/W and UNSAT-H predict minimal percolation, whereas the predicted percolation using HYDRUS-2D is about half of that based on LEACHM. In the case of UNSAT-H, the over prediction in SRO dominates the water balance such that percolation is significantly under predicted, whereas in the case of VADOSE/W, the over prediction in ET appears to be the primary reason for the under prediction in percolation.

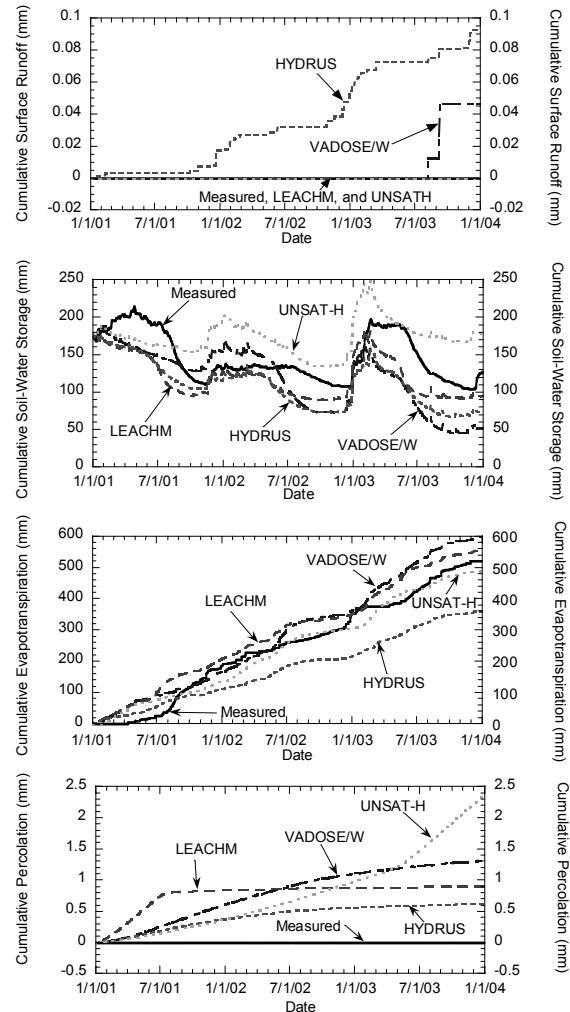


Figure 33. Measured versus predicted water balance parameters based on four numerical unsaturated flow models for the monolithic cover constructed at Boardman, OR.

In the case of HYDRUS-2D, the relatively accurate predictions in both SRO and ET suggest that the under prediction in percolation may be due to the differences in scales for the different water balance parameters. For example, at the end of the records based on HYDRUS-2D in Fig. 32, almost 100 mm of SRO is predicted, and there is almost 100 mm of SWS. However, although the predicted ET based on HYDRUS-2D appears accurate relative to the field ET, close scrutiny at the end of the record indicates that the predicted ET of approximately 800 mm is approximately 50 mm greater than the measured ET of approximately 750 mm. This difference in ET of about 50 mm is more than sufficient to make up for the slight difference between the predicted and measured percolations of less than 4 mm based on HYDRUS-2D. A similar explanation can be made in the case of VADOSE/W, where the predicted ET at the end of the record is approximately 200 mm ( $\approx 950 \text{ mm} - 750 \text{ mm}$ ) greater than the measured ET.

Unlike the Altamont site, all four models tend to over predict percolation at the Boardman site (Fig. 33). The SRO at Boardman predicted by VADOSE/W and LEACHM does not appear to be accurate, although the overall amount of SRO for the period of record is extremely low ( $< 0.1 \text{ mm}$ ), especially in comparison with the SRO at Altamont (Fig. 32). As a result, SRO does not play a significant role at the Boardman site. Thus, the over predictions in percolation by the models may be due to the use of a unit gradient for the lower boundary condition (LBC), which allows for more water to flow out of the lower boundary than may be permitted in the field. Although a seepage face

LBC typically is used to simulate unsaturated flow through alternative covers (Scanlon et al., 2002), a unit gradient LBC was used for all simulations reported herein because UNSAT-H is capable of handling only a unit gradient boundary condition.

While the comparisons between the measured and predicted water balances for alternative covers shown in Figs. 32 and 33 represent only a small fraction of those being conducted in the study, these comparisons do serve as examples of some of the problems inherent in trying to predict the field performance of engineered systems, particularly when the processes involved are highly non linear and the properties of the systems are time dependent. These examples also illustrate why evaluation and verification of numerical models will continue to be an important issue facing geotechnical engineers, particularly within the realm of environmental issues.

## 8 IMPORTANCE OF PROFESSIONAL IDENTITY

The plethora of terminology that has been used to represent the general field of environmental issues related to Geotechnical Engineering (e.g., waste geotechnics, environmental geotechnology, environmental control, Geotechnical Engineering for waste disposal, geotechnical practice for waste disposal, geotechnics of waste disposal, environmental geotechnics, geoenvironmental engineering, etc.) has, in some respects, led to confusion and an overall lack of formal professional identity for individuals addressing such issues. While some of the terminology undoubtedly refers to a specific aspect of the general field (e.g., waste disposal), other terminology is more broad-based in meaning or interpretation. Also, the exact origin of the terminology is rarely known with any certainty and, therefore, the inferred meaning of the terminology may not be exactly the same as the intended meaning.

Regardless of the seeming variability and/or vagueness in terminology, two terms, viz. "Environmental Geotechnics" and "Geoenvironmental Engineering" (or "Geo-Environmental Engineering"), appear to have gained some consensus through an evolutionary process for providing a suitable identity for those who work in the general field of environmental issues related to Geotechnical Engineering. While the difference in these two terms also may be difficult to discern, the writer believes there is a distinction in meaning between the two terms. In the viewpoint of the writer, the term "Geoenvironmental Engineering" is the more broad-based term reflecting the multidisciplinary aspects of soil-environmental problems from an engineering perspective, whereas the term "Environmental Geotechnics" is a more specific term referring primarily to those aspects of soil-environmental problems from the perspective of Geotechnical Engineering.

For example, many professionals such as hydrogeologists, groundwater engineers, environmental engineers, geotechnical engineers, and others, may be involved with environmental problems related to subsurface pollution. However, each of these professionals probably will be involved in different aspects of a specific problem, such as contaminant migration (hydrogeologist), well-head protection (groundwater engineer), chemical and physical treatment of waste streams (environmental engineers), and design and construction of waste containment systems (geotechnical engineers). From a broad or multidisciplinary perspective, each of these individuals is a "geoenvironmental engineer or scientist" because each individual is dealing with some aspect of subsurface pollution. However, the expertise of each individual within the general field of "Geoenvironmental Engineering and Science" obviously is not the same.

At the same time, each individual in this multidisciplinary viewpoint may have been formally educated in a different field. For example, a hydrogeologist is a geologist who specializes in the origin, age, and movement of subsurface water. This spe-

cialization distinguishes the hydrogeologist from other geologists who have essentially the same basic education in geology, but may have a more in-depth knowledge of some other aspect of geology, such as geomorphology. Therefore, hydrogeology refers to a sub-discipline of geology. In a similar manner, groundwater engineers deal with problems not related to the environment (e.g., well hydraulics and aquifer production), Environmental Engineering includes many other environmental topics not related to soils (e.g., air pollution, surface water pollution, design of sanitary treatment facilities, etc.), and Geotechnical Engineering encompasses many other diverse areas related to the design and construction of civil works (e.g., shallow and deep foundations, soil dynamics, slope stability analyses, and earth retaining structures).

From the perspective of Geotechnical Engineering, a geotechnical engineer whose primary responsibility is the design and construction of foundations may be considered to be a "foundation engineer". In a similar manner, a geotechnical engineer whose primary responsibility pertains to design and construction of clay liners and covers for waste disposal may be associated with the sub-discipline of "Environmental Geotechnics". While each individual within Geotechnical Engineering is a geotechnical engineer and, therefore, is educated to some minimum level in the essential subjects associated within Geotechnical Engineering (e.g., soil mechanics and soil behavior, foundations), the expertise of each geotechnical engineer is not the same. Therefore, from the perspective of Geotechnical Engineering, the sub-discipline of Geotechnical Engineering whose primary focus relates to environmental issues can be referred to as "Environmental Geotechnics" (see Fig. 34).

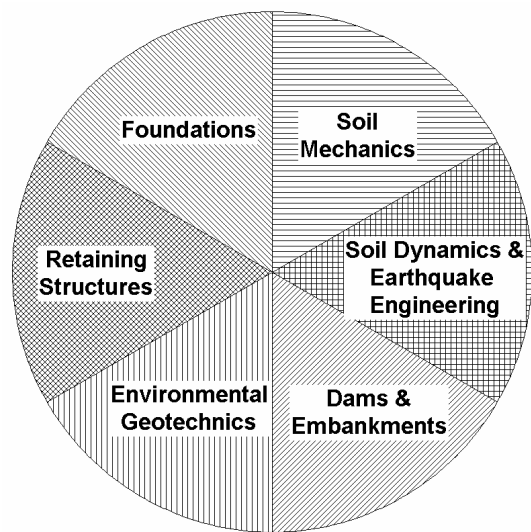


Figure 34. The geotechnical sub-disciplinary pie.

However, based on the above discussion, there may be a perception that the specializations of individuals who work in Geoenvironmental Engineering and Science are the sole domain of those individuals. On the contrary, all geoenvironmental engineers and scientists should work to achieve a minimum level of knowledge in as many of the other specialized areas in Geoenvironmental Engineering and Science as possible. The attainment of this breadth of knowledge will help to provide for the effective technical interaction between professionals that is required for successful completion of geoenvironmental projects in a multidisciplinary setting. For example, individuals who are considered to be specialists in Environmental Geotechnics not only should possess the minimum level of knowledge which characterizes a geotechnical engineer but also should possess a minimum level of knowledge to effectively interact with hydrogeologists, groundwater engineers, environmental engineers,

etc. The attainment of this knowledge will require the learning of subjects that generally are considered outside the scope of Geotechnical Engineering, such as contaminant transport, soil chemistry, aqueous chemistry, and soil microbiology.

The Geoenvironmental Engineering program at Colorado State University is based largely on these concepts and consists of an interdisciplinary program represented by the intersection of three sub-disciplines within Civil Engineering, viz. Geotechnical Engineering, Environmental Engineering, and Groundwater Engineering (see Fig. 35). While this definition may seem somewhat restricted, the definition does facilitate the ability of students in the Geoenvironmental Engineering program to originate from any one of the three primary sub-disciplines of Civil Engineering, and to complement their formal education in their chosen sub-discipline by taking courses in the other two sub-disciplines as well as a number of other disciplines related to geoenvironmental science (e.g., hydrogeology, soil science, etc.).

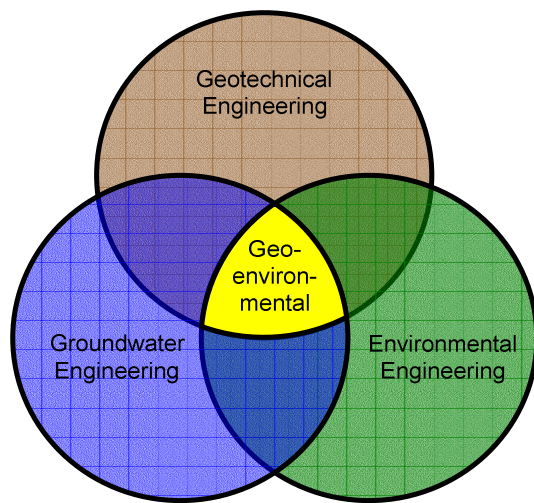


Figure 35. Definition of the geoenvironmental engineering program at Colorado State University ([www.engr.colostate.edu/ce/grad/geoenv.shtml](http://www.engr.colostate.edu/ce/grad/geoenv.shtml)).

As an example of the potential significance of the issue of terminology and professional identity, an ongoing debate has existed within the Geotechnical Engineering community in the United States since the name of the flagship professional journal published by the American Society of Civil Engineers was changed from the *Journal of Geotechnical Engineering* to the *Journal of Geotechnical and Geoenvironmental Engineering* beginning with the first issue (Vol. 123, No. 1) published in 1997. Some individuals have argued that the name of the journal should not have been changed because Geoenvironmental Engineering represents a sub-discipline of Geotechnical Engineering, suggesting that the other sub-disciplines of Geotechnical Engineering are of lesser significance and importance, and that the change in the name of the journal has made the journal more exclusive rather than more inclusive (e.g., Mitchell, 2002). However, based on the aforementioned distinction between the terms Geoenvironmental Engineering and Environmental Geotechnics, the term Geoenvironmental Engineering is not a sub-discipline of Geotechnical Engineering, but rather represents the intersection of all the engineering fields that are involved in Geoenvironmental Engineering applications. Thus, one could argue that the change in the name of the journal has expanded, rather than contracted, the scope and intended audience of the journal. If this argument is accepted, then the Geotechnical Engineering community has taken a leading role in terms of formalizing the identity of Geoenvironmental Engineering.

In summary, an individual whose primary education is that of a geotechnical engineer but who specializes in the solution to environmental problems related to geo-materials may be identi-

fied from the perspective of the Geotechnical Engineering profession as specializing in the area of "Environmental Geotechnics". At the same time, the same individual may be identified as a "geoenvironmental engineer" from the perspective of any other professional whose primary education is in a discipline other than Geotechnical Engineering, but who also specializes in the solution to environmental problems related to soils (e.g., hydrogeologist, groundwater engineer, environmental engineer, etc.). However, a "geoenvironmental engineer or scientist" whose primary formal education is not in Geotechnical Engineering would not be considered to work in the area of "Environmental Geotechnics".

## 9 SUMMARY

An assessment of seven current and/or future environmental issues in Geotechnical Engineering has been presented, including the

- (1) long-term performance of existing waste containment systems,
- (2) acceptance of alternative barriers and barrier systems,
- (3) need for innovative barriers and barrier systems,
- (4) emergence of new forms of waste,
- (5) increasing importance of biological processes,
- (6) role of modeling, and
- (7) need for consistent terminology with respect to professional identity.

The significance and practical application of each issue was illustrated through one or more examples.

Two examples were provided to illustrate the need for continued monitoring of waste containment systems over long durations to ensure that such systems remain effective in protecting human health and the environment. The first example was a description of the recent appearance of volatile organic compounds (VOCs) in lysimeters beneath lined landfill cells in Wisconsin, USA. The second example pertained to the role of cation exchange on the hydraulic performance of geosynthetic clay liners (GCLs) and the relatively long durations that can be required to reach equilibrium in the hydraulic conductivity of GCLs permeated with inorganic salt solutions.

With respect to alternative barriers and barrier materials, emphasis was placed on alternative earthen final covers (AEFCs) that are currently being considered as substitutes for more conventional or prescriptive final covers at waste containment facilities. Advantages of AEFCs include lower costs and potentially better resistance to environmental distress factors such as desiccation cracking and freeze-thaw cycles. The results of water balance measurements of three AEFCs at three different sites within the United States were presented to emphasize the need for large-scale field testing in order to evaluate the performance of AEFCs relative to conventional covers, and the criteria currently being used to establish equivalent performance of AEFCs relative to conventional covers based on the U. S. EPA's Alternative Cover Assessment Program (ACAP) were presented.

Innovative barriers and barrier materials are distinguished from alternative barriers and barrier materials in that the former are still in a fundamental stage of research and development whereas the latter are more in the demonstration stage for regulatory approval and application. Two possible innovative barriers and barrier materials were described, including clay barriers that behave as semi-permeable membranes, or clay membrane barriers (CMBs), and polymer-clay nanocomposites. The primary advantage of CMBs is a reduction in the contaminant (solute) mass flux relative to non-membrane clay barriers. However, the ability of CMBs to sustain membrane behavior over prolonged periods is an issue requiring additional study. Polymer-clay nanocomposites refer to plastic (polymer) materials that contain a small fraction of layered silicate clay minerals

resulting in improved material properties for the nanocomposite relative to the polymer alone. In particular, the strength of polymer-clay nanocomposites can be as much as two times greater than that for the polymer alone, and gaseous diffusion coefficients for polymer-clay nanocomposites can be substantially reduced relative to those for the polymer alone. Due to these improvements in properties, polymer-clay nanocomposites may offer enhanced performance as substitutes for geomembranes in waste containment systems, although such enhanced performance has not been demonstrated to date.

In order to protect human health and the environment, geotechnical engineers will have to deal with the issue of the fate and mobility of several potential emerging waste forms that either recently have been identified or are expected in the near future. Three such waste forms were identified, including antibiotics used for growth promotion in livestock (cattle, pigs, poultry, sheep), prions resulting from the disposal of animal carcasses infected with animal transmitted diseases, such as transmissible spongiform encephalopathies (TSEs), in waste containment facilities, and nanowaste resulting as by-products of the growing nanotechnology industry.

The increasing importance of biological processes in governing the performance of environmental systems also is an emerging issue in Geotechnical Engineering. The role of biological processes in three environmental applications involving Geotechnical Engineering was described to illustrate this increasing significance, including bioreactor landfills, bioremediation of organic compounds via biostimulation or bioaugmentation with electrokinetics, and treatment of metal laden ground waters using sulfate reducing bacteria in passive treatment systems, such as permeable reactive trenches and engineered wetlands. The complexity of biological processes relative to physical and chemical processes represents a significant challenge to geotechnical engineers, such that the role of biological processes in Geotechnical Engineering likely will remain a dominant environmental issue for the foreseeable future.

The role of modeling is a particularly important issue when dealing with environmental problems because models often are used as tools to predict the future impacts and resulting risk from engineering activities related to environmental protection, such as new waste disposal facilities and existing sources of contamination (e.g., waste dumps and piles, accidental chemical spills, etc.). Because the uncertainty in the accuracy of the model predictions can be considerable, predictions made with existing models generally cannot be considered reliable until and unless the predictions are verified by comparison with field data. In this regard, the results of the predicted water balances based on four commercially available unsaturated flow models are compared with the field measured water balances for two AEFCs at two sites in the United States. The trends in the measured water balance parameters generally are mimicked similarly well for all four models, but the abilities of each model to accurately predict a given water balance parameter varies among the four models and between the two sites. The comparison serves to illustrate that although progress is being made in terms of determining the reliability of models to predict future outcomes, considerable study is still required such that the role of modeling likely will continue to be an issue in terms of environmental applications in Geotechnical Engineering.

Finally, an argument is made for the need for consistent terminology with respect to the professional identity of individuals who deal with environmental issues. The writer proposes the use of two terms with different meanings, viz., Environmental Geotechnics and Geoenvironmental Engineering. In this regard, Environmental Geotechnics refers to the sub-discipline of Geotechnical Engineering that includes individuals whose primary education is that of a geotechnical engineer but who specialize in the solutions to environmental problems related to geomaterials. In contrast, Geoenvironmental Engineering is a more broad-based term reflecting the multidisciplinary aspects of

soil-environmental problems from an engineering perspective, and includes individuals whose primary training may be in disciplines other than Geotechnical Engineering, such as environmental engineers and groundwater engineers. Consistency in such terminology should help to provide a consistency in professional identity that can be more accurately and efficiently conveyed to the general public at large.

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