Contaminant Transport through GML/CCL Bottom Liner with Consideration of Consolidation Effects

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ABSTRACT
This paper presents a numerical investigation of the effect of compacted clay liner (CCL) consolidation on contaminant transport through a composite bottom liner system consisting of a geomembrane (GML) overlying a CCL. Numerical simulations were conducted using the CST2 model, which accounts for one-dimensional coupled large strain consolidation and contaminant transport in saturated porous media. Simulation results indicate that CCL consolidation can have a significant effect on contaminant transport through a composite GML/CCL bottom liner system, not only during the course of consolidation but also after the consolidation process has finished. Analyses based on diffusive transport alone neglect consideration of transient advection and changes of CCL properties caused by consolidation and can result in significant errors.

1. INTRODUCTION

Composite liner systems consisting of a geomembrane (GML) overlying and in intimate contact with a compacted clay liner (CCL) are used at the base of solid waste (e.g., sanitary) landfills to isolate waste materials and landfill leachate from the surrounding environment. Contaminant transport analyses for such systems are traditionally performed using advective-diffusive models that assume the CCL is rigid and ignore transient advection and associated changes in material properties due to CCL consolidation (Foose 2002). In reality, waste filling operations can extend over many years and apply high vertical stresses to bottom liner systems that cause CCL consolidation which, in turn, can affect contaminant transport. The applied loading generates excess pore water pressures that increase advective transport, whereas the consolidation process reduces the void ratio of the clay and decreases the thickness of the CCL over time. The hydraulic conductivity and effective diffusion coefficient of the clay also decrease. For example, Rowe (2005) reported that the average hydraulic conductivity of the CCL in the bottom liner system at the Keele Valley Landfill decreased by more than one order of magnitude as a result of waste filling operations. Field studies reported by Workman (1993) and Othman et al. (1997) indicated that contaminant breakthrough occurred much earlier than theoretical predictions based on diffusive transport, and such earlier than predicted transport has been attributed to consolidation effects (Othman et al. 1997, Rowe 1998). However, the consideration of consolidation-induced transport for landfill bottom liner systems has not received much attention until recently (Peters and Smith 2002, Fox 2007b, Lewis et al. 2009, Zhang et al. 2013).

This paper presents a numerical investigation of the effects of CCL consolidation on contaminant transport through a composite GML/CCL bottom liner system using realistic geometry and material properties. More detailed analyses and results are presented by Pu et al. (2014). Numerical simulations were conducted using the validated CST2 numerical model that accounts for coupled large strain consolidation-induced transport (Fox and Lee 2008). The CST2 model is briefly described, followed by a comparison of numerical simulation results for two analysis methods to highlight the effects of consolidation. Errors associated with traditional advective-diffusive analysis also are discussed.

2. NUMERICAL MODELS

CST2 (Consolidation and Solute Transport 2) is a numerical model for the simulation of coupled large strain consolidation and contaminant transport in saturated soil. The geometry for the CST2 model is shown in Figure 1. A homogeneous, saturated soil layer of initial height H_o is treated as an idealized two-phase material in which the solid particles and pore fluid are incompressible. The solid phase is represented as a column of elements. The vertical coordinate and solid element coordinate are defined as positive upward (against gravity) from a fixed datum at the base of the layer. The pore fluid is also represented as a column of elements. Each fluid element has an initial fluid concentration and initial dissolved solute mass. The initial sorbed (solid-phase) concentration for each solid element is assumed to be in equilibrium with the local fluid concentration.

Developed using the CS2 method (Fox and Berles 1997), the consolidation algorithm for CST2 is one-dimensional and accounts for vertical strain, soil self-weight, general constitutive relationships, relative velocity of fluid and solid phases, changing hydraulic conductivity and compressibility during consolidation, time-dependent loading, unloading/reloading effects, and an external hydraulic gradient. Soil constitutive relationships are defined using discrete points and can take nearly any desired form. The contaminant transport algorithm accounts for advection, diffusion, mechanical dispersion, equilibrium or nonequilibrium (i.e., kinetic) sorption, linear or nonlinear sorption, and a soil porosity-dependent effective diffusion coefficient. Contaminant transport is consistent with temporal and spatial variations of porosity and seepage velocity in the consolidating soil. The key to the transport algorithm is the definition of two Lagrangian fields of elements that separately follow the motions of fluid and solid phases. This approach reduces numerical dispersion and simplifies transport calculations to those of dispersive mass flow between contiguous fluid elements (Fox 2007a). The top and bottom boundary conditions with respect to solute transport can be specified as prescribed concentration (Type I), concentration gradient (Type II), or solute mass flux (Type III). CST2 can also accommodate a reservoir boundary, which represents an accumulating well-mixed aqueous reservoir formed by fluid outflow at the top boundary. Details regarding the development of CST2 are provided by Fox (2007a,b), and Fox and Lee (2008).

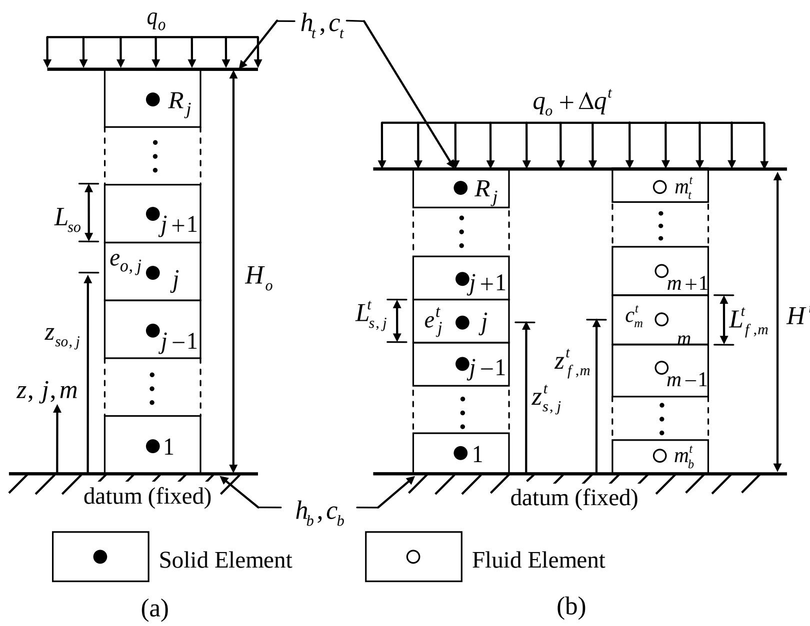

Figure 1. Geometry for CST2: (a) initial configuration; (b) configuration during loading (Fox and Lee 2008).

3. NUMERICAL SIMULATIONS

CST2 was used to evaluate the effects of consolidation on contaminant transport for a composite GML/CCL bottom liner system. The initial geometry is shown in Figure 2. The liner system consists of, from top to bottom, a leachate collection system (LCS), a 1.5-mm-thick intact high density polyethylene (HDPE) GML, a 1-m-thick CCL, and subgrade layer. The subgrade can represent an underlying leachate detection layer or a natural soil subgrade. The top boundary of the CCL is undrained due to presence of the GML. The bottom boundary of the CCL is drained and at constant atmospheric pressure. The GML carries a constant head of leachate of 0.3 m in the overlying LCS. The leachate contains a volatile organic compound, trichloroethylene (TCE), with constant concentration, c_o, of 100 mg/L and is assumed to be sufficiently dilute so as to not alter the properties of the CCL. The CCL is initially uncontaminated and has a constant, zero concentration at the base. This bottom concentration condition (c = 0) allows for diffusion across the interface that results in the highest (i.e., most conservative) estimate of contaminant mass flux. The TCE undergoes diffusive transport through the GML prior to reaching the CCL, and then undergoes advective-dispersive transport through the CCL with sorption according to the organic carbon content of the clay. Diffusive transport through the GML was treated using the method described by Fox (2007b). TCE transport parameters for the GML are taken from experimental investigations reported by Sangam and Rowe (2001), with a GML diffusion coefficient, D_{GML}, of 4×10^{-13} m^2/s and partition coefficient, K_{GML}, of 85. TCE transport parameters for the CCL are taken from experimental investigations reported by Kim et al. (2001), Boving and Grathwohl (2001), Charbeneau (2000), and Mercer and Cohen (1990).

CCL compressibility and hydraulic conductivity constitutive relationships are taken from Fox (2007b) and correspond to an incremental-loading consolidation test conducted on a lean clay (USCS classification = CL, liquid limit = 41, plastic limit = 20) used for CCL construction at a municipal solid waste landfill in the Midwest U.S. Hydraulic conductivity was measured after each load increment using a syringe flow pump. The constitutive relationships display trends similar to natural soils, including a nearly constant compression index and hydraulic conductivity that is primarily related to void ratio during unloading/reloading. The hydraulic conductivity relationship can be expressed as e = 2.53 + 0.198logk, where e = void ratio and k = vertical hydraulic conductivity (m/s). The effective diffusion coefficient, D*, for TCE diffusion in the CCL varies with solid element porosity as $D^* = D_o (n)^M$, where D_o = free solution diffusion coefficient, n = soil porosity, and M = effective diffusion coefficient exponent (Lerman 1978). Based on values reported by Kim et al. (2001) and Boving and Grathwohl (2001), $D_o = 8.6\times10^{-10}$ m^2/s and M = 1. TCE sorption is characterized using a linear equilibrium isotherm with distribution coefficient, K_d, expressed as $K_d = K_{oc} f_{oc}$ (Charbeneau 2000), where K_{oc} = organic carbon partition coefficient, which is 126 mL/g for TCE (Mercer and Cohen 1990), and f_{oc} = mass fraction of organic carbon contained within the CCL. Decay of TCE (e.g., due to biological processes) is neglected. The key transport parameters are summarized in Table 1.

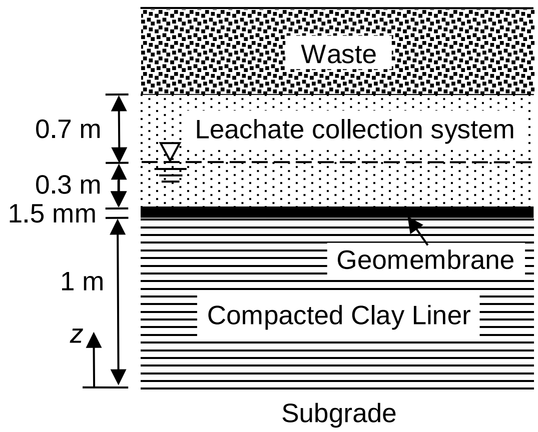

Figure 2. Initial geometry for GML/CCL liner system.

Table 1. TCE transport parameters for the CCL.

Property	CCL with no organic carbon	CCL with organic carbon
Free solution diffusion coefficient D_o (m^2/s)	8.6×10^{-10} [a]	8.6×10^{-10} [a]
D* exponent for porosity M	1[b]	1[b]
Longitudinal dispersivity α_L (m)	0.02[a]	0.02[a]
Organic carbon content f_{oc} (%)	0	0.5
Distribution coefficient K_d (mL/g)	0	0.63[c]

[a]Kim et al. 2001
[b]Boving and Grathwohl 2001
[c]Mercer and Cohen 1990, Charbeneau 2000

Initial stress conditions are calculated assuming the CCL is saturated and initially in hydraulic equilibrium (i.e., no flow), with a uniform total head distribution of h = 0 taken with respect to elevation z = 0, prior to waste placement. The LCS layer is 1-m-thick (constant) with saturated unit weight γ_{sat} = 20.6 kN/m^3 for the lower 0.3 m and moist unit weight γ = 17.5 kN/m^3 for the upper 0.7 m. Under these conditions and neglecting the weight of the GML, the initial effective stress at the top of the CCL, σ_o' (= $0.7\gamma + 0.3\gamma_{sat} + H_o\gamma_w$, where γ_w = unit weight of water), is 28.2 kPa. Starting at time t = 0, waste placement causes an increase in vertical stress at an assumed constant rate of 100 kPa/yr for a total loading period of 10 years, giving a final applied stress Δq = 1000 kPa. The total elapsed time for each simulation is 50 years. These conditions correspond to a municipal solid waste landfill with a final waste height of approximately 70 to 90 m, a 10-year filling period, and a 40-year post-closure period. Although transport parameters are constant for each numerical simulation, spatial and temporal changes in porosity and seepage velocity produce variations of hydrodynamic dispersion, retardation factor, and Peclet number during consolidation (Pu 2014). Simulations were conducted using 120 solid elements, 120 fluid elements for simulations without sorption, and 360 fluid elements for simulations with sorption. These levels of numerical resolution should yield high accuracy to the simulation results (Fox 2007b).

4. SIMULATION RESULTS

Plots of applied stress (Δq), settlement (S), and bottom boundary fluid outflow rate (v) for the composite GML/CCL bottom liner system are presented in Figure 3. The applied stress increased linearly to a final value of 1000 kPa at 10 years and remained constant thereafter. Consolidation was completed at about t = 10.5 yr and the resulting final settlement was 94 mm, giving a final average strain of 9.4%. Based on the average void ratio, CCL hydraulic conductivity decreased 84% (from 2.80×10^{-10} m/s to 4.64×10^{-11} m/s) and D* decreased 16% (from 3.35×10^{-10} m^2/s to 2.80×10^{-10} m^2/s) over the same time period. The fluid outflow rate at the bottom boundary was relatively high at the start of loading, gradually decreased as excess pore pressures dissipated, and then decreased rapidly after the cessation of loading (t = 10 yr). At t = 10.5 yr, dissipation of excess pore pressures was essentially complete such that the fluid outflow rate reached zero.

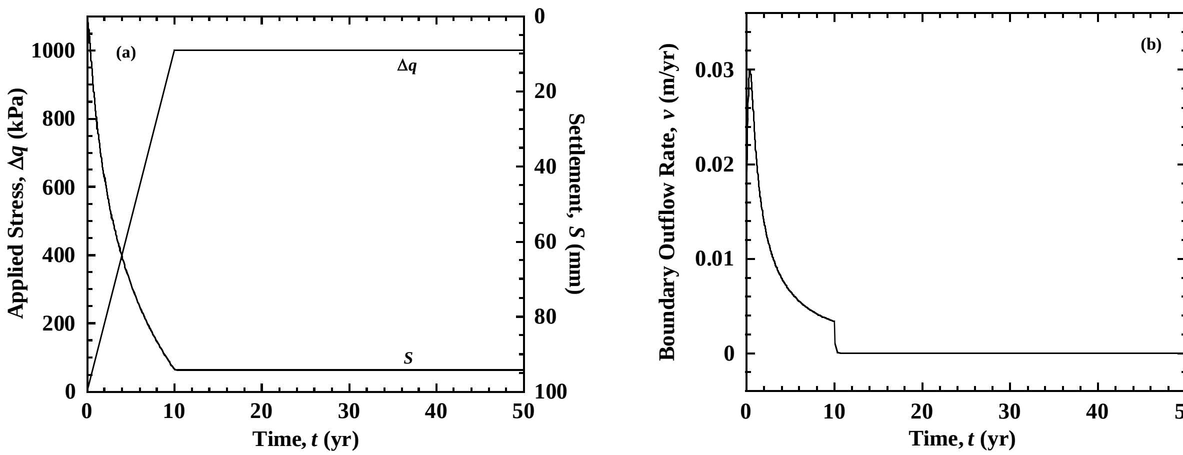

Figure 3. Consolidation response: (a) applied stress and settlement; (b) bottom boundary fluid outflow rate.

Simulations for TCE transport through the GML/CCL liner system were conducted for two cases. The case "C" simulations corresponded to consolidation-induced transport followed by diffusive transport after completion of consolidation. The case "DI" simulations neglected consolidation effects and corresponded to diffusive transport using initial conditions for the liner system (i.e., constant initial CCL thickness and constant initial profiles of void ratio and effective diffusion coefficient) over the entire 50-yr evaluation period. Results for both cases C and DI were obtained using CST2 with appropriate input parameters (Pu 2014). Simulation results for contaminant mass flux (F) at the bottom boundary of the CCL are shown in Figure 4(a). For a CCL without sorption ($K_d = 0$), results for the C case indicated initial TCE breakthrough in approximately 3 years. Thereafter, mass flux increased to approximately 0.3 mg/m^2/yr and then started to level off at approximately 0.315 mg/m^2/yr as steady state conditions were approached near the end of the evaluation period. From t = 10 to 10.5 yr, there was a temporary, minor decrease in flux, because loading stopped and, as a result, advective transport began to decrease sharply (see Figure 3b). In contrast, the case DI simulation for $K_d = 0$ predicted slightly later breakthrough (~ 3.5 yr) and lower mass flux in the early stages due to the lack of advective transport, but higher mass flux soon after the end of consolidation, because D* remained at the initial (higher) value and was not reduced by consolidation of the clay. For a CCL with sorption ($K_d > 0$), the case C simulation indicated TCE initial breakthrough occurred at approximately 12 years, which represented a significant delay relative to the 3-yr breakthrough for the no-sorption case. At the end of the 50-yr evaluation period, mass flux for the DI case was again higher than that for the C case; however, both cases indicated substantially lower flux relative to that resulting from the simulations for a CCL without sorption.

Corresponding plots of normalized mass flux, or F/F_{ss}, are shown in Figure 4(b), where F_{ss} is the steady-state mass flux for a given condition. Values of F_{ss} were calculated using the analytical solution of Foose et al. (2002) and were equal to 0.317 mg/m^2/yr for case C (using final CCL thickness and final profiles of e and D*) and 0.410 mg/m^2/yr for case DI (using initial CCL thickness and initial profiles of e and D*). The results indicate that, without sorption, case C yielded higher F/F_{ss} values throughout the evaluation period, and that steady state mass flux was nearly achieved for both cases C and DI at t = 50 yr. In contrast, for a sorbing CCL, values of F/F_{ss} were lower for case C relative to case DI throughout the evaluation period, and were significantly lower than the steady state value (1.0) at the end of the evaluation period.

The effect of CCL consolidation on contaminant transport is indicated more clearly in Figure 5, which shows the ratio R of mass flux for case C divided by that for case DI. In preparing this plot, R was truncated when the mass flux value for either case was lower than 1×10^{-7} mg/m^2/yr. R for a non-sorbing CCL decreased from approximately 10 at t = 1.5 yr to

0.78 at t = 50 yr, which indicates that the case DI simulation underestimated mass flux by about one order of magnitude at the beginning of loading and overestimated mass flux by 28% at the end of the evaluation period. R was greater than unity during the consolidation stage, because advection increased the magnitude of downward transport which had a greater overall effect relative to the concurrent decrease in D*. Soon after the end of consolidation, R became less than unity because advection ceased and D* and e were reduced by consolidation of the clay. For a sorbing CCL, R ranged from 0.4 to 0.6, indicating that mass flux for case DI was approximately twice that obtained from a full consolidation-induced transport analysis. These R values are lower than unity because, with transport delayed by sorption, consolidation reduced D* before TCE reached the bottom boundary of the CCL for the case C simulation.

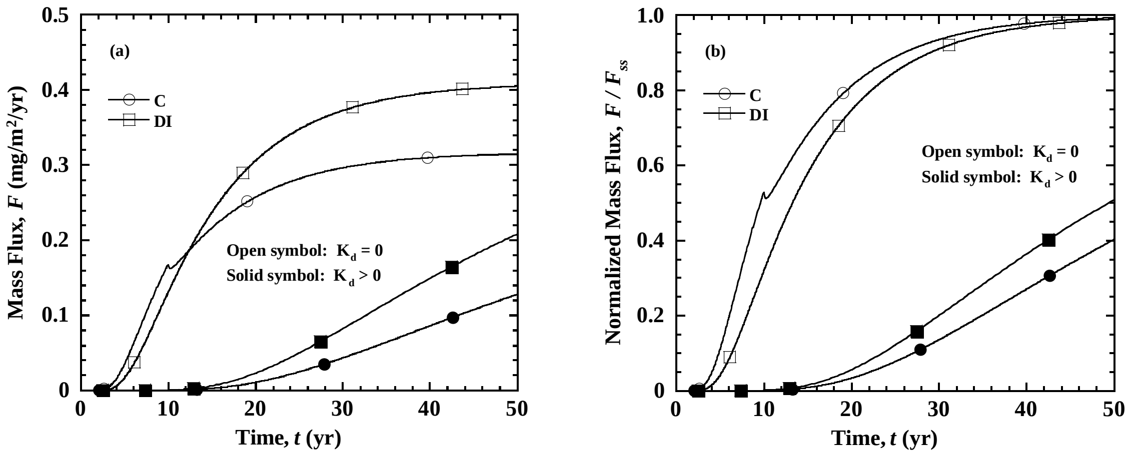

Figure 4. Simulation results for bottom boundary of CCL: (a) contaminant mass flux; (b) normalized contaminant mass flux.

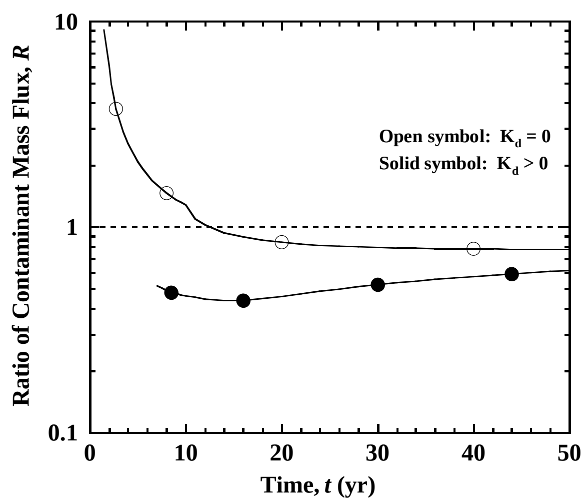

Figure 5. Ratio of contaminant mass flux at bottom boundary of CCL.

5. CONCLUSIONS

This paper presented the results of a numerical investigation of the effect of consolidation of a compacted clay liner (CCL) on contaminant transport through a composite GML/CCL bottom liner system. Numerical simulations were conducted using the CST2 model, which accounts for one-dimensional, coupled large strain consolidation and contaminant transport in saturated porous media. Simulation results indicated that CCL consolidation can have a significant effect on contaminant transport through a composite GML/CCL bottom liner system, not only during the course of consolidation but also after the consolidation process has finished. Analyses based on diffusive transport alone neglect transient advection and changes of CCL properties caused by consolidation and can lead to significant errors.

ACKNOWLEDGEMENTS

Financial support for this investigation was provided by Grant No. CMMI-1001023 and Grant No. CMMI-0969346 from the Geotechnical Engineering Program of the U.S. National Science Foundation. This support is gratefully acknowledged.

REFERENCES

Boving, T.B., and Grathwohl, P. (2001). Tracer diffusion coefficients in sedimentary rocks: correlation to porosity and hydraulic conductivity. *Journal of Contaminant Hydrology*, 53, 85-100.

Charbeneau, R.J. (2000). *Groundwater Hydraulics and Pollutant Transport*. Prentice-Hall, Inc., Upper Saddle River, NJ.

Foose, G.J. (2002). Transit-time design for diffusion through composite liners. *Journal of Geotechnical and Geoenvironmental Engineering*, 128, 590-601.

Foose, G.J., Benson, C.H., and Edil, T.B. (2002). Comparison of solute transport in three composite liners. *Journal of Geotechnical and Geoenvironmental Engineering,* 128, 391-403.

Fox, P.J. (2007a). Coupled large strain consolidation and solute transport. I: Model development. *Journal of Geotechnical and Geoenvironmental Engineering,* 133, 3-15.

Fox, P.J. (2007b). Coupled large strain consolidation and solute transport. II: Model verification and simulation results. *Journal of Geotechnical and Geoenvironmental Engineering*, 133, 16-29.

Fox, P.J., and Berles, J.D. (1997). CS2: A piecewise-linear model for large strain consolidation. *International Journal for Numerical and Analytical Methods in Geomechanics*, 21, 453-475.

Fox, P.J., and Lee, J. (2008). Model for consolidation-induced solute transport with nonlinear and nonequilibrium sorption. *International Journal of Geomechanics*, 8, 188-198.

Kim, J.Y., Edil, T.B., and Park, J.K. (2001). Volatile organic compound (VOC) transport through compacted clay. *Journal of Geotechnical and Geoenvironmental Engineering*, 127, 126-134.

Lerman, A. (1978). Chemical exchange across sediment-water interface. *Annual Review of Earth and Planetary Sciences*, 6, 281-303.

Lewis, T.W., Pivonka, P., and Smith, D.W. (2009). Theoretical investigation of the effects of consolidation on contaminant transport through clay barriers. *International Journal for Numerical and Analytical Methods in Geomechanics*, 33, 95-116.

Mercer, J.W., and Cohen, R.M. (1990). A review of immiscible fluids in the subsurface: Properties, models, characterization and remediation. *Journal of Contaminant Hydrology*, 6, 107-163.

Othman, M.A., Bonaparte, R., and Gross, B.A. (1997). Preliminary results of composite liner field performance study. *Geotextiles and Geomembranes*, 15, 289-312.

Peters, G.P., and Smith, D.W. (2002). Solute transport through a deforming porous medium. *International Journal for Numerical and Analytical Methods in Geomechanics*, 26, 683-717.

Pu, H. (2014). *Computational Investigations of Large Strain Consolidation and Consolidation-Induced Contaminant Transport*. Ph.D. Thesis, University of California-San Diego, La Jolla, CA.

Pu, H., Fox, P.J., and Shackelford, C.D. (2014). Assessment of consolidation-induced contaminant transport for geoenvironmental applications. I: Compacted clay liner systems. *Journal of Geotechnical and Geoenvironmental Engineering*, in preparation.

Rowe, R.K. (1998). Geosynthetics and the minimization of contaminant migration through barrier systems beneath solid waste. *Proceedings of the 6th International Conference on Geosynthetics*, Atlanta, March, Vol. 1, pp. 27-103, Industrial Fabrics Association International, St. Paul, MN.

Rowe, R.K. (2005). Long-term performance of contaminant barrier systems. *Geotechnique*, 55, 631-678.

Sangam, H.P., and Rowe, R.K. (2001). Migration of dilute aqueous organic pollutants through HDPE geomembranes. *Geotextiles and Geomembranes*, 19, 329-357.

Workman, J.P. (1993). Interpretation of leakage rates in double lined systems. *Proceedings of the 7th GRI Conference: Geosynthetic Liner Systems*, Philadelphia, 91-108.

Zhang, H.J., Jeng, D.-S., Barry, D.A., Seymour, B.R., and Li, L. (2013). Solute transport in nearly saturated porous media under landfill clay liners: A finite deformation approach. *Journal of Hydrology*, 479, 189-199.