

# HYDRAULIC CONDUCTIVITY OF A GEOSYNTHETIC CLAY LINER TO A SIMULATED ANIMAL WASTE SOLUTION

L. C. Brown, C. D. Shackelford

**ABSTRACT.** *The potential use of a commercially available geosynthetic clay liner (GCL) as the primary hydraulic barrier for animal waste lagoons was evaluated on the basis of hydraulic conductivity ( $k$ ) testing. The GCL was permeated under both aerobic and anaerobic conditions with both deionized water (DIW) and a simulated animal waste solution to determine the effect, if any, of the simulated solution on  $k$  of the GCL relative to  $k$  based on DIW. The chemical composition of the simulated solution was based on reported compositions for animal waste lagoons in Colorado. The  $k$  of the GCL increased only slightly (8%) on average when permeated with the simulated solution under anaerobic conditions relative to the  $k$  of  $1.3 \times 10^{-9}$  cm/s based on permeation with DIW, but increased by a factor of 4.2 on average when permeated with the simulated solution under aerobic conditions. The measured  $k$  values are shown to be consistent with those from other studies involving permeation of GCLs with various chemical solutions at the same ionic strength as that for the simulated animal waste solution used in this study. The practical implications of the results relative to regulations governing liners for animal waste lagoons in 12 states are discussed.*

**Keywords.** *Anaerobic animal waste lagoons, Bentonite, CAFO, Geosynthetic clay liners, Hydraulic conductivity.*

One of the major changes in animal agriculture during the past few decades has been the replacement of small to mid-size animal operations with larger concentrated animal feeding operations (CAFOs), defined as animal feeding operations with more than 1,000 animal units or beef feeder equivalents (U.S. EPA, 2003). The improper management of the waste produced by these CAFOs has the potential to contaminate ground water and surface water (Miller et al., 1985; Waggoner et al., 1995; Ham and DeSutter, 1999). As a result, the common practice is to store animal wastes in anaerobic lagoons or storage basins that are lined with up to 0.91 m (3 ft) of a low-permeability ( $\leq 10^{-7}$  cm/s) compacted clay liner (CCL) (Ritter and Chirnside, 1990; Ham and DeSutter, 2000; Reddi and Davalos, 2000). The purpose of the CCL is to prevent excessive seepage from the lagoon and thereby eliminate or minimize groundwater and/or surface water contamination.

At present, the design standards for animal waste lagoons vary by state (Parker et al., 1999; Reddi and Davalos, 2000). For example, while some states regulate the maximum allowable hydraulic conductivity ( $k$ ), other states regulate the maximum infiltration (seepage) rate and/or the minimum

liner thickness (e.g., Parker et al., 1999). This variability in design standards results, in part, from the variability in soil conditions and properties that inevitably exist from state to state as well as within any given state. In this regard, more uniform design standards may be possible if a more uniform liner material, such as manufactured geosynthetic clay liners, could be used in place of a CCL.

Geosynthetic clay liners (GCLs) are factory-manufactured rolls consisting of thin layers (~10 mm) of sodium bentonite (~5.0 kg/m<sup>2</sup>) sandwiched between two geotextiles or bonded to a geomembrane by a chemical adhesive. The structural integrity of GCLs is maintained by stitching or needle-punching through the geotextiles and bentonite, and/or by adding an adhesive compound to the bentonite to bind the bentonite to the geotextile or geomembrane (polymer sheet). The geotextiles offer essentially no resistance to flow, but rather serve to contain the bentonite within the manufactured product and to provide the frictional resistance to sliding along the interfaces between the geotextiles and the surrounding materials when the GCLs are placed on the side slopes of waste disposal facilities (Daniel et al., 1998). The low  $k$  that is commonly obtained when GCLs are permeated with water (i.e., typically  $k \leq 3 \times 10^{-9}$  cm/s) is due to the bentonite, which swells in the presence of water, resulting in constriction of the flow paths between the clay particles or particle aggregates (Daniel et al., 1997).

The low  $k$  of GCLs, the relative ease of installation, and their low cost relative to CCLs all make the potential use of GCLs for lining animal waste lagoons highly advantageous. However, several studies have shown that the concentration and type of solutes in the permeant liquid play an important role in determining the hydraulic conductivity of bentonite-based barriers. For example, James et al. (1997) found cracking in the bentonite of GCLs used to cap underground water reservoirs, resulting in unacceptable seepage through

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Submitted for review in July 2005 as manuscript number SW 5998; approved for publication by the Soil & Water Division of ASABE in February 2007.

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the GCLs. A subsequent investigation revealed that the cracking resulted from shrinkage of the bentonite when calcium ( $\text{Ca}^{2+}$ ) exchanged with sodium ( $\text{Na}^+$ ) originally on the exchange complex of the bentonite. Similar results were reported by Melchior (1997), who investigated the cause for excessive leakage from a GCL-lined test section of a landfill cover in Germany.

These findings of James et al. (1997) and Melchior (1997) are also consistent with several studies that have indicated significant increases in the hydraulic conductivity of sand-bentonite mixtures and GCLs upon permeation with calcium-dominated solutions (Shan and Daniel, 1991; Stern and Shackelford, 1998; Shackelford et al., 2000; Kolstad et al., 2004; Jo et al., 2005; Lee and Shackelford, 2005). For example, Lee and Shackelford (2005) reported increases in  $k$  of greater than two orders of magnitude (i.e., relative to  $k$  based on permeation with water), resulting in final  $k$  values  $>10^{-7}$  cm/s (i.e., greater than the maximum hydraulic conductivity typically allowed for waste containment liners) for specimens of two GCLs permeated with solutions containing  $\geq 50$  mM  $\text{CaCl}_2$  or  $\geq 100$  mM  $\text{CaCl}_2$ , depending on the quality of the bentonite in the GCL. Such hydraulic conductivity tests involving non-standard permeant liquids (i.e., permeant liquids other than water) often are referred to as compatibility tests, because the primary objective of the test is to evaluate the compatibility between the permeant liquid and the soil.

In general, the permeant liquids used for compatibility tests performed with GCLs have involved simple monovalent salt solutions, such as sodium chloride ( $\text{NaCl}$ ) solutions (e.g., Petrov and Rowe, 1997; Petrov et al., 1997a, 1997b), simple divalent salt solutions, principally calcium chloride ( $\text{CaCl}_2$ ) solutions (e.g., Shan and Daniel, 1991; Shackelford et al., 2000; Lee and Shackelford, 2005), or solutions containing mixtures of monovalent and divalent salts (e.g., Kolstad et al., 2004; Jo et al., 2005). However, no compatibility tests have been reported for GCLs permeated with either actual or simulated animal waste streams used as the permeant liquids. As a result, the objective of this study was to evaluate the potential for a GCL as a liner or liner component for animal waste lagoons in terms of the ability of the GCL to maintain a suitably low  $k$  upon permeation with a simulated animal waste stream.

## MATERIALS AND METHODS

### GEOSYNTHETIC CLAY LINER

The GCL used in this study consists of a layer of bentonite sandwiched between woven and non-woven geotextiles (e.g., Shackelford et al., 2000). The geotextiles are needle-punched together such that the resulting thickness of the GCL in the air-dried state is approximately 10 mm. Since the GCL did not include a geomembrane component, the hydraulic performance of the GCL was based on the swelling behavior of the bentonite.

The physical and chemical properties and the mineralogical composition of the bentonite portion of the GCL are summarized in table 1. As indicated in table 1, the bentonite was dominated by montmorillonite, the primary swelling mineral in bentonite that results in the low hydraulic conductivity (Shackelford et al., 2000). In addition, the dominance of sodium ( $\text{Na}^+$ ) on the exchange sites of the

**Table 1. Physical and chemical properties and mineralogical compositions of the bentonite in the geosynthetic clay liner (GCL) evaluated in this study.**

Property	Standard	Value
Specific gravity	ASTM D 854	2.78
Liquid limit, LL	ASTM D 4318	376
Plasticity index, PI	ASTM D 4318	336
Soil classification	ASTM D 2487	CH
Swell index, SI (mL/2 g)	ASTM D 5890	32
Bentonite mass per unit area ( $\text{kg}/\text{m}^2$ )	ASTM D 5993	4.6
Principal minerals (%):	[a]	
Montmorillonite		76
Illite/mica		1
K feldspar		3
Plagioclase feldspar		7
Quartz		11
Other		2
Cation exchange capacity, CEC ( $\text{cmol}_c/\text{kg}$ )	[b]	75.3
Exchangeable metals ( $\text{cmol}_c/\text{kg}$ ):	[b]	
$\text{Ca}^{2+}$		9.1
$\text{Mg}^{2+}$		3.8
$\text{Na}^+$		55.9
$\text{K}^+$		8.7
Total		77.5
Soluble metals ( $\text{mg}/\text{kg}$ ):	[b, c]	
Ca		2846
Mg		36
Na		7448
K		117
Sodium adsorption ratio, SAR	[b, c]	12
Soil pH	[b, d]	8.6
Soil electrical conductivity, EC ( $\text{mS}/\text{m}$ ) at $25^\circ\text{C}$	[b, d]	440

[a] Based on X-ray diffraction analyses performed by Mineralogy, Inc., Tulsa, Oklahoma.

[b] Based on procedures described in Shackelford and Redmond (1995).

[c] Measured from a 1 g:20 mL clay:water extract.

[d] Measured on a saturated soil paste.

bentonite confirms that the bentonite was a sodium bentonite. The sum of the total exchangeable metals of  $77.5 \text{ cmol}_c/\text{kg}$  is within 3% of the measured cation exchange capacity (CEC) of  $75.3 \text{ cmol}_c/\text{kg}$ , with the slight difference likely being due to measurement uncertainty.

The results of a mechanical sieve analysis (ASTM D 422) indicated that the air-dried bentonite was granular such that the bentonite would classify as poorly graded sand (SP) according to the Unified Soil Classification System, or USCS (ASTM D 2487) in an air-dried condition. However, the results of a hydrometer analysis (ASTM D 422) indicated that 75% (w/w) of the bentonite was comprised of clay-sized particles (i.e.,  $75\% < 5 \mu\text{m}$ ). Based on the results from the hydrometer analysis, and the liquid limit and the plasticity index given in table 1, the bentonite classifies as high-plasticity clay (CH) according to the USCS.

### PERMEANT LIQUIDS

The permeant liquids used in this study consisted of tap water that was processed by passage through three Barnstead ion exchange columns in series (Barnstead International, Dubuque, Iowa), and a simulated animal waste solution. The compositions and properties of these permeant liquids are given in table 2. Based on the measured electrical conductiv-

**Table 2. Compositions and properties of permeant liquids used in this study.**

Parameter	Permeant Liquid	
	Deionized Water	Simulated Animal Waste Solution
Electrical conductivity, EC (mS/m) at 25°C	0.20	606
pH	5.60	6.10
Dissolved oxygen, DO (mg/L)	7.80	7.80
Redox potential, Eh (mV)	191	51.5
NO <sub>3</sub> <sup>-</sup> -N (mg/L)	0	0
NH <sub>4</sub> <sup>+</sup> -N (mg/L)	0	379
Cl <sup>-</sup> (mg/L)	0	1697
Ca <sup>2+</sup> (mg/L)	0	139
Na <sup>+</sup> (mg/L)	0	502
PO <sub>4</sub> <sup>3-</sup> (mg/L)	0	150
Ionic strength, I (M)	~ 0	0.059

ity (EC) at 25°C of 0.2 mS/m, the processed tap water classifies as Type IV deionized water (DIW) in accordance with ASTM D 1193 (EC < 0.5 mS/m). The simulated solution was prepared by dissolving anhydrous inorganic salts (CaCl<sub>2</sub>, NaCl, NH<sub>4</sub>Cl, and NaHCO<sub>3</sub>) and phosphoric acid (H<sub>3</sub>PO<sub>4</sub>) in the DIW. The salts and the H<sub>3</sub>PO<sub>4</sub> were obtained from Sigma-Aldrich (St. Louis, Mo.) with purities greater than 99%.

In formulating the chemical composition of the simulated animal waste solution, consideration was given primarily to the compositions for actual animal wastes reported for Colorado, as shown in table 3. A comparison of the chemical compositions for the simulated animal waste solution in table 2 versus those for the actual animal wastes in table 3 reveals that the concentrations of the primary species in the simulated solution are generally conservative (high) with respect to swine, dairy, and cattle lagoons in Colorado. In addition, the NH<sub>4</sub><sup>+</sup>-N concentration of 379 mg/L is higher

than that reported by Reddi and Davalos (2000) for a swine lagoon in Kansas, and the Cl<sup>-</sup> concentration of 1697 mg/L is substantially higher than any of the values reported by the CDC (1998) for swine lagoons in Iowa. Originally, a lower Cl<sup>-</sup> concentration was used in the solution and bicarbonate (HCO<sub>3</sub><sup>-</sup>) was used as an anion in the solution, but the high bicarbonate concentrations led to precipitation of calcium carbonate (CaCO<sub>3</sub>(s)). To remedy this situation, the concentration of HCO<sub>3</sub><sup>-</sup> was reduced by substitution with Cl<sup>-</sup>, resulting in the relatively high concentration of 1697 mg/L for Cl<sup>-</sup>.

#### HYDRAULIC CONDUCTIVITY

Specimens of the GCL with diameters of 102 mm were permeated in flexible-wall permeameters using the constant-head procedure (ASTM D 5084). The flexible-wall permeameter was chosen because this permeameter provides the greatest assurance that sidewall leakage does not occur (Lin and Benson, 2000; Shackelford et al., 2000).

The hydraulic conductivity (*k*) measurements of the GCL were conducted using both permeant liquids (i.e., DIW and the simulated animal waste solution) under both aerobic and anaerobic conditions with two testing apparatuses that differed slightly depending on testing condition (see Brown, 2004, for detailed schematics). The primary difference between the two apparatuses pertained to the use of bladder accumulators for the tests conducted under anaerobic conditions. Bladder accumulators are often used in compatibility testing to isolate the permeant liquid from the surrounding atmosphere (Daniel, 1994; Shackelford, 1994).

Anaerobic conditions were created by diffusing nitrogen gas through the simulated solution until the dissolved oxygen (DO) level, as measured with a DO meter (Oakton D100, Oakton Instruments, Inc., Vernon Hills, Ill.) was less than 4.0 mg/L, and subsequently transferring the solution to the bladder accumulator without exposure to the atmosphere.

**Table 3. Chemical properties of animal wastes.**

Reference	Location	Date	Type	Species and Concentration (mg/L)							
				NH <sub>4</sub> <sup>+</sup> -N	NO <sub>3</sub> <sup>-</sup> -N	P	K <sup>+</sup>	Na <sup>+</sup>	Ca <sup>2+</sup>	Mg <sup>2+</sup>	Cl <sup>-</sup>
Miller et al. (1976)	Ontario	Spring 1974	Swine	541	82	0.18	--	--	--	--	--
Miller et al. (1985)	Ontario	Fall 1975	Cattle	2610	0	--	2808	3243	480	243	--
				2680	--	420	1080	--	480	170	--
				1450	--	630	690	--	590	230	--
				2770	--	660	1470	--	500	260	--
				3500	--	930	1770	--	670	310	--
				4280	--	810	2000	--	870	250	--
				4550	--	930	2040	--	800	330	--
				5330	--	960	2970	--	780	260	--
Campbell et al. (1997)	Prince Edward Island	Fall 1992	Swine	5670	--	800	1900	--	1140	360	--
				1400	9.3	180	--	--	--	--	380
				650	0.26	45	--	--	--	--	300
				860	0	78	--	--	--	--	240
				2000	0	510	--	--	--	--	680
				620	0	55	--	--	--	--	260
				1400	0	130	--	--	--	--	390
CDC (1998)	Iowa	Oct. to Dec. 1998	Swine	1800	0	500	--	--	--	--	760
				235	--	--	--	--	--	--	1020
				325	1	20	334	200	88	59	253
				200-420	<1-8	60	500	--	--	--	--
				270	--	--	--	--	--	--	--
Reddi and Davalos (2000)	Kansas	1999	Swine	235	--	--	--	--	--	--	
Jepson (2002)	Colorado	2001	Swine	325	1	20	334	200	88	59	253
		2001	Dairy	200-420	<1-8	60	500	--	--	--	--
		2001	Cattle	270	--	--	--	--	--	--	--

Masters (1997) indicates that the saturated value of DO in water ranges from about 8 to 15 mg/L, depending on temperature and salinity, and that anaerobic conditions become prevalent as the DO drops below about 4 or 5 mg/L. Thus, anaerobic conditions were assumed to exist for DO below 4.0 mg/L.

The specimens were back-pressure saturated with the permeant liquid using air either without (aerobic condition) or with (anaerobic condition) bladder accumulators. The final cell pressure after back-pressure saturation was 276 kPa (40 psi), such that the average effective stress of the specimen was 34.5 kPa (5 psi).

After saturation, permeation was induced by maintaining a differential hydraulic head of 4.22 m (13.85 ft) across the GCL, resulting in a hydraulic gradient of 422 (i.e., 4.22 m/0.01 m). Although a hydraulic gradient of 422 is significantly higher than the maximum gradient of 30 stipulated in ASTM D 5084 for soils with  $k \leq 10^{-7}$  cm/s, hydraulic gradients ranging from 50 to 600 typically are used for measuring the  $k$  of GCLs due to the low  $k$  typically measured for GCLs (e.g., Shan and Daniel, 1991; Daniel et al., 1993; Didier and Comeaga, 1997; Petrov and Rowe, 1997; Petrov et al., 1997a, 1997b; Quaranta et al., 1997; Ruhl and Daniel, 1997; Lin and Benson, 2000; Shackelford et al., 2000; Jo et al., 2005; Lee and Shackelford, 2005; Lee et al., 2005). In addition, the results of an analysis performed by Shackelford et al. (2000) indicate that the  $k$  of GCLs is affected to a greater extent by average effective stress than by the magnitude of hydraulic gradient.

The tests using DIW as the permeant liquid were not terminated before the following criteria specified in ASTM D 5084 were achieved: (1) four consecutive  $k$  values within  $\pm 50\%$  of the mean  $k$  value for the four measurements if  $k < 1 \times 10^{-10}$  m/s ( $1 \times 10^{-8}$  cm/s), or within  $\pm 25\%$  if the measured  $k$  value was  $> 1 \times 10^{-10}$  m/s; (2) four consecutive values of the ratio of outflow to inflow equal to  $1.00 \pm 0.25$ ; and (3) a minimum of two pore volumes of flow (PVF) of liquid permeated through the specimens. The termination criteria for the hydraulic conductivity tests using the simulated animal waste solution as the permeant liquid included those for the tests using DIW as well as the requirements based on ASTM D 6766 that the ratios of outflow-to-inflow pH and EC are within  $1.00 \pm 0.10$ . These additional termination criteria in ASTM D 6766 are required to ensure that chemical equilibrium between the outflow and inflow essentially has been established such that all chemical interactions between the permeant liquid and the soil have been completed (e.g., Daniel, 1994; Shackelford, 1994; Shackelford et al., 1999, 2000; Jo et al., 2005; Lee and Shackelford, 2005). Satisfaction of all termination criteria for the compatibility tests required between about 42 and 104 days of permeation.

For the compatibility tests, samples of both the inflow and outflow were collected periodically inside a cell saturated with nitrogen to prevent the intrusion of oxygen. The samples were immediately monitored for both pH using a pH meter (Accumet AB 15, Fisher Scientific, Inc., Fair Lawn, N.J.) and EC using a conductivity meter (Thermo Orion 150 A+, Thermo Electric, Inc., Beverly, Mass.) to evaluate the aforementioned termination criterion for establishment of chemical equilibrium between the inflow and outflow. The samples also were monitored for dissolved oxygen (DO) using the DO meter and redox potential (Eh) using the pH

meter. Dissolved oxygen was measured to evaluate the effectiveness of maintaining aerobic or anaerobic conditions during the tests based on the 4.0 mg/L criterion. The Eh was monitored to evaluate the effect, if any, of Eh on the hydraulic conductivity of the GCL (e.g., Shen et al., 1992; Kamon et al., 2002).

In the case of low-permeability materials, such as GCLs, that are permeated with chemical solutions requiring establishment of chemical equilibrium prior to termination, greater than duplicate replication of testing has rarely, if ever, been performed primarily due to the resulting long testing durations and the common limitation in the number of available testing apparatuses (e.g., Jo et al., 2001, 2005; Kolstad et al., 2004; Lee and Shackelford, 2005; Lee et al., 2005). In such cases, the reliability of the results has been based more on comparison with the results reported for other similar materials and methods as opposed to any statistical comparison based on multiple replicates. This same approach was followed in the current study, in which all tests were duplicated and the results were compared with those based on tests involving similar materials and methods.

## RESULTS

### PERMEATION WITH DIW

Figure 1 shows the data from the duplicate hydraulic conductivity ( $k$ ) tests using DIW as the permeant liquid, under aerobic and anaerobic conditions, versus time and pore volumes of flow (PVF), and table 4 summarizes the final  $k$  values for the tests. All four tests resulted in  $k$  values between  $1.2 \times 10^{-9}$  and  $1.5 \times 10^{-9}$  cm/s. This relatively narrow range of  $k$  values is consistent with the 99th percentile  $k$  value of  $1.3 \times 10^{-9}$  cm/s reported by Daniel et al. (1997) based on the results of a round-robin testing program involving 18 laboratories using flexible-wall permeability cells to permeate a GCL with DIW. In addition, the geometric means of the  $k$  values based on the two different test conditions are reasonably close, with less than 10% difference between the geometric mean  $k$  of  $1.4 \times 10^{-9}$  cm/s for the duplicate tests performed under aerobic conditions and the geometric mean  $k$  of  $1.3 \times 10^{-9}$  cm/s for the duplicate tests performed under anaerobic conditions. Thus, the condition under which the GCL was permeated with DIW had no practical effect on the resulting hydraulic conductivity for the GCL.

### PERMEATION WITH SIMULATED ANIMAL WASTE SOLUTION

#### *Hydraulic Conductivity*

The data for the duplicate hydraulic conductivity ( $k$ ) tests using the simulated animal waste solution as the permeant liquid versus time and PVF under aerobic and anaerobic conditions are shown in figures 2 and 3, respectively, and the resulting  $k$  values are summarized in table 4. Chemical equilibrium between outflow and inflow based both on pH (i.e.,  $\text{pH}_{\text{out}}/\text{pH}_{\text{in}} = 1.00 \pm 0.10$ ) and EC (i.e.,  $\text{EC}_{\text{out}}/\text{EC}_{\text{in}} = 1.00 \pm 0.10$ ) was obtained in  $\leq 16$  PVF for both tests conducted under aerobic conditions (fig. 2) and  $\leq 15$  PVF for both tests conducted under anaerobic conditions (fig. 3). Thus, regardless of the test condition (i.e., aerobic or anaerobic), the durations of all four compatibility tests in terms of the PVF required to establish chemical equilibrium were nearly the same.

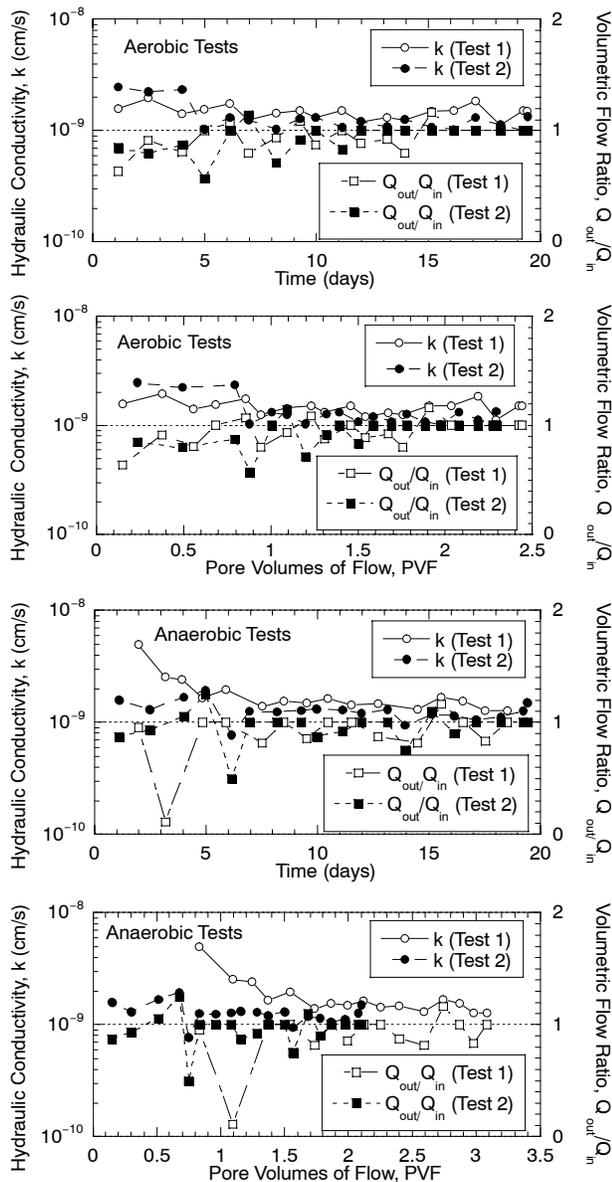


Figure 1. Hydraulic conductivity results based on permeation of duplicate GCL specimens with deionized water.

Table 4. Summary of results of hydraulic conductivity tests conducted in this study.

Permeant Liquid	Test Condition	Hydraulic Conductivity, $k$ (cm/s)		
		Test	Measured	Geometric Mean
Deionized water	Aerobic	1	$1.5 \times 10^{-9}$	$1.3 \times 10^{-9}$
		2	$1.2 \times 10^{-9}$	
	Anaerobic	1	$1.4 \times 10^{-9}$	$1.3 \times 10^{-9}$
		2	$1.2 \times 10^{-9}$	
Simulated animal waste solution	Aerobic	1	$4.0 \times 10^{-9}$	$5.5 \times 10^{-9}$
		2	$7.7 \times 10^{-9}$	
	Anaerobic	1	$1.5 \times 10^{-9}$	$1.4 \times 10^{-9}$
		2	$1.4 \times 10^{-9}$	

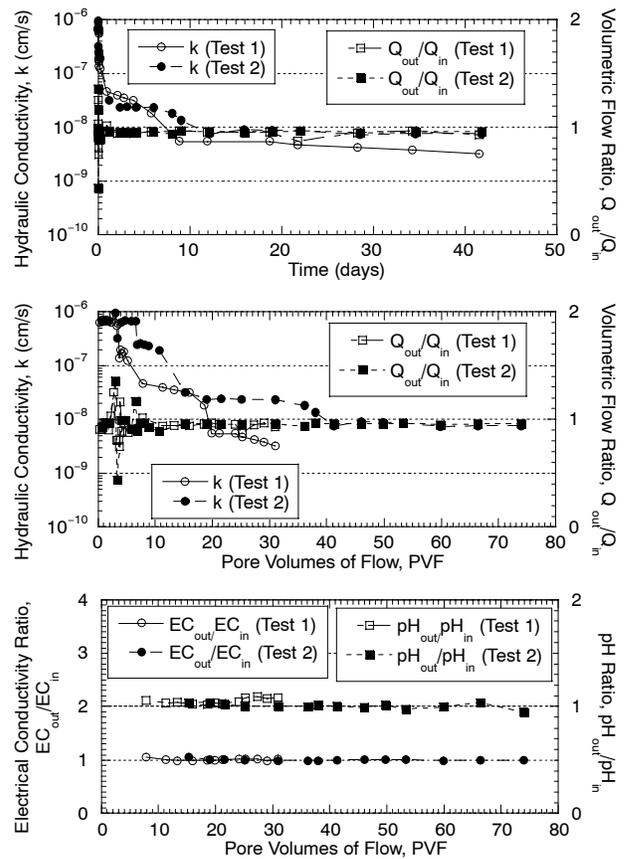


Figure 2. Hydraulic conductivity data based on permeation of duplicate GCL specimens with simulated animal waste solution under aerobic conditions.

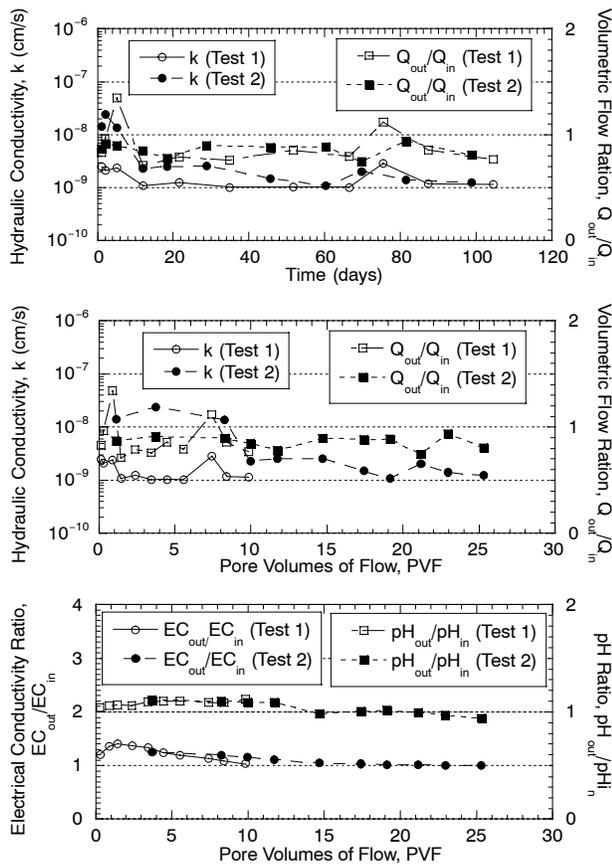
### Dissolved Oxygen

The measured dissolved oxygen (DO) concentrations in the inflows and outflows for all four compatibility tests are shown in figure 4 as a function of PVF. Except for a relatively brief period ( $20 < \text{PVF} < 26$ ) when the measured DO concentrations in the outflow of test 1 conducted under aerobic conditions decreased almost to zero, the measured DO concentrations in both the inflow and outflow of the tests conducted under aerobic conditions consistently range from about 10 to 15 mg/L, indicating that aerobic conditions (i.e., base

on  $\text{DO} > 4.0$  mg/L) were maintained throughout the duration of the majority of these tests. In contrast, the inflow and outflow DO concentrations for the tests conducted under anaerobic conditions never exceeded 4.0 mg/L, indicating that anaerobic conditions were maintained throughout the durations of these tests.

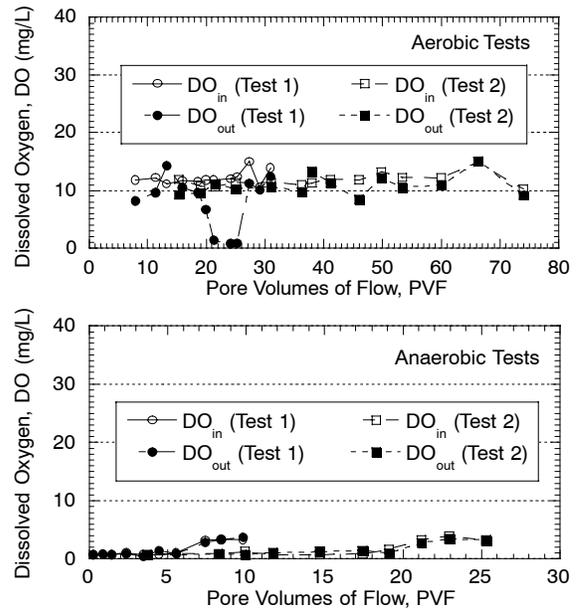
### Redox Potential

The measured redox potentials (Eh and pe), where  $\text{Eh (mV)} = 59\text{pe}$  (Benjamin, 2002), in the inflows and outflows of all four compatibility tests are shown in figure 5. Three observations can be made from the data shown in figure 5. First, the ranges of redox potentials in the outflows from the two tests performed under aerobic conditions (test 1:  $-4 \text{ mV} \leq \text{Eh}_{\text{out}} \leq 22 \text{ mV}$ ; test 2:  $20 \text{ mV} \leq \text{Eh}_{\text{out}} \leq 48 \text{ mV}$ ) are generally lower than the ranges of redox potentials in the outflows from the two tests performed under anaerobic



**Figure 3. Hydraulic conductivity data based on permeation of duplicate GCL specimens with simulated animal waste solution under anaerobic conditions.**

conditions (test 1:  $35 \text{ mV} \leq E_{h_{out}} \leq 65 \text{ mV}$ ; test 2:  $28 \text{ mV} \leq E_{h_{out}} \leq 88 \text{ mV}$ ). Thus, although most of the measured redox potentials in the outflows for all four compatibility tests are positive values, indicating that all four specimens were generally under oxidizing conditions, the specimens tested under aerobic conditions generally were more reduced relative to the specimens tested under anaerobic conditions. The reasons for this difference are unknown, particularly since aerobic conditions were expected to result in a greater



**Figure 4. Dissolved oxygen concentrations in the inflow and outflow of duplicate GCL specimens permeated with simulated animal waste solution under both aerobic and anaerobic conditions.**

tendency for oxidation relative to anaerobic conditions. However, no attempt was made in this study to control any biological activity within the specimens (e.g., via use of a biocide). As a result, the observed difference in the range of redox potentials between tests performed under aerobic conditions relative to tests performed under anaerobic conditions may be due, in part, to differences in biological activity.

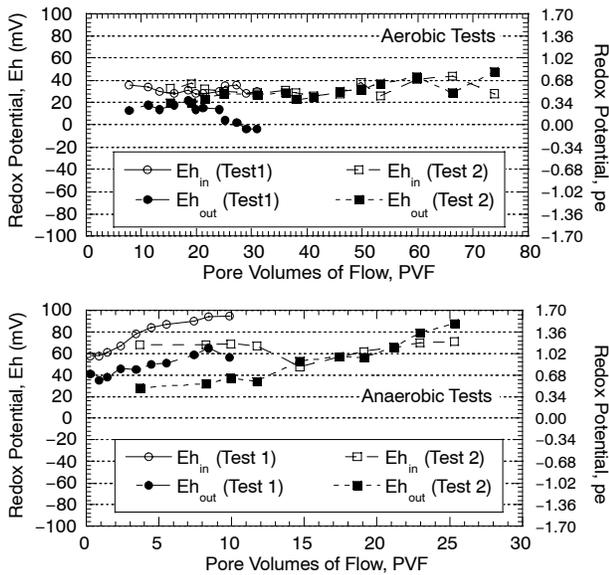
Second, the redox potentials in the outflows from one of the specimens tested under aerobic conditions (test 2) and both specimens tested under anaerobic conditions increased with PVF (i.e., time), indicating a trend towards increasing oxidation with continued permeation in these three tests. However, the redox potentials in the outflow from the other specimen tested under aerobic conditions (test 1) tended to decrease with time such that the system transitioned from an oxidized state at the beginning of the test to a reduced state at the time of termination. The reason for the decrease in

**Table 5. Technical properties and infiltration requirements for earthen liners used for animal waste lagoons and storage ponds based on state regulations (data from Parker et al. 1999).**

State	Minimum Liner Thickness (cm)	Maximum Hydraulic Conductivity, $k$		Maximum Infiltration Rate, $I$	
		(cm/d)	(cm/s)	(cm/d)	(cm/s)
Colorado (CO)	30.5	--	--	0.08	$9.3 \times 10^{-7}$
Idaho (ID)	--	--	--	0.86	$1.0 \times 10^{-5}$
Iowa (IA)	--	--	--	0.16	$1.9 \times 10^{-6}$
Kansas (KS)	15	--	--	0.063	$7.3 \times 10^{-7}$
Missouri (MO)	30.5	--	--	0.042 or 0.29	$4.9 \times 10^{-7}$ or $3.4 \times 10^{-6}$
Nebraska (NE)	--	--	--	0.63	$7.3 \times 10^{-6}$
New Mexico (NM)	30.5	0.0086	$10^{-7}$	--	--
North Carolina (NC)	--	0.0864 or 0.0086	$10^{-6}$ or $10^{-7}$	--	--
Oklahoma (OK)	45.7	0.0086	$10^{-7}$	--	--
South Dakota (SD)	--	0.0086	$10^{-7}$	--	--
Texas (TX) <sup>[a]</sup>	--	0.0086	$10^{-7}$	--	--
Wisconsin (WI) <sup>[b]</sup>	61 (sides), 46 (bottom)	0.0086	$10^{-7}$	--	--

[a] Subchapter K regulations.

[b] Option 1 regulations.



**Figure 5. Redox potentials in inflow and outflow of duplicate GCL specimens permeated with simulated animal waste solution under both aerobic and anaerobic conditions.**

$E_{h_{out}}$  with PVF for this specimen is not known, particularly since exposure of the permeant liquid to air would be expected to induce increasing oxidation, as opposed to reduction, and the test durations for the two specimens tested under aerobic conditions were essentially the same (fig. 2). However, the previously noted decrease in DO in the outflow for this specimen (i.e., test 1) to almost zero between about 20 and 26 PVF (fig. 4), combined with the tendency towards reducing conditions, suggests that biological activity in this specimen may have been greater than that in the other specimen tested under aerobic conditions.

Third, for the two specimens tested under anaerobic conditions, the redox potentials measured in the outflow ( $E_{h_{out}}$ ) were generally less than those measured in the inflow ( $E_{h_{in}}$ ), i.e., except after 15 PVF for test 2. An  $E_{h_{out}} < E_{h_{in}}$  indicates a tendency towards reducing conditions within the specimen during permeation, which is expected for the tests performed under anaerobic conditions. In contrast,  $E_{h_{out}}$  was expected to be similar to  $E_{h_{in}}$  for the tests performed under aerobic conditions. While this expectation was met in the case of test 2,  $E_{h_{out}}$  was less than  $E_{h_{in}}$  for test 1. Although unexpected,  $E_{h_{out}} < E_{h_{in}}$  for test 2 conducted under aerobic conditions is consistent with the previously noted anomalous behavior for this specimen with respect to DO concentrations in the outflow and the tendency towards reducing conditions upon continued permeation (i.e., decreasing  $E_{h_{out}}$  with time), which may be due to elevated biological activity in this test, as previously noted.

## DISCUSSION

### EVALUATION OF MEASURED HYDRAULIC CONDUCTIVITY

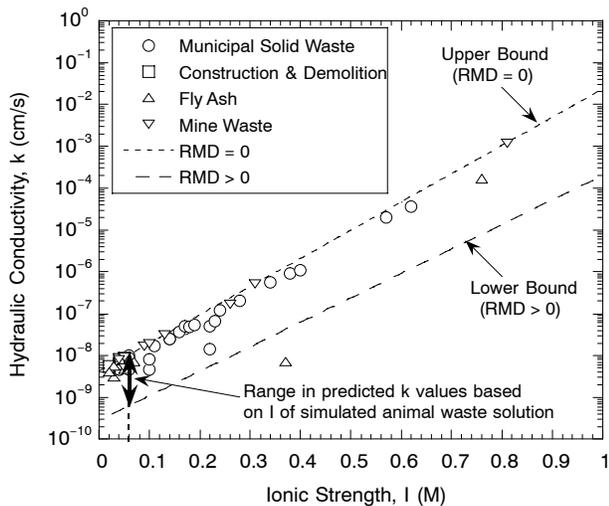
As shown in table 4, the  $k$  values of the duplicate specimens of the GCL permeated under anaerobic conditions with the simulated animal waste solution were nearly identical, with a geometric mean  $k$  of  $1.4 \times 10^{-9}$  cm/s. This mean  $k$  is virtually the same as the geometric mean  $k$  of  $1.3 \times 10^{-9}$  cm/s based on permeation of specimens of the GCL with

DIW under aerobic and anaerobic conditions. In contrast, the  $k$  of the duplicate specimens of the GCL permeated under aerobic conditions with the simulated animal waste solution are noticeably different, with the  $k$  from test 2 of  $7.7 \times 10^{-9}$  cm/s being about twice the  $k$  of  $4.0 \times 10^{-9}$  cm/s for test 1, resulting in a geometric mean  $k$  of  $5.5 \times 10^{-9}$  cm/s. This geometric mean  $k$  is 3.9 times greater than that based on permeation with the simulated solution under anaerobic conditions, and 4.2 times greater than the geometric mean  $k$  based on permeation with DIW under aerobic or anaerobic conditions. Thus, some incompatibility between the simulated solution and the bentonite in the GCL resulted when the GCL was permeated under aerobic conditions.

However, the magnitude of the resulting increase in  $k$  is significantly lower relative to the aforementioned significant increases ( $\geq 100$  times) that have been observed when some GCLs have been permeated with some chemical solutions containing divalent metals (e.g.,  $CaCl_2$ ). For example, the measured concentration of  $Ca^{2+}$  in the simulated animal waste solution of 139 mg/L corresponds to only 3.5 mM  $CaCl_2$ . This concentration is lower than the lowest concentration of 5 mM  $CaCl_2$  used by Lee and Shackelford (2005), which resulted in an increase in  $k$  relative to DIW of 3.6 times for the GCL containing bentonite with a similar amount of montmorillonite (i.e., 77% vs. 76% for the GCL bentonite in this study).

Finally, Kolstad (2000) and Kolstad et al. (2004) correlated the  $k$  of GCLs permeated with inorganic chemical solutions with two chemical parameters: (1) the ionic strength of the solution (I), and (2) the ratio of the molar concentrations of monovalent cations ( $M_M$ ) to divalent cations ( $M_D$ ) in the solution, or RMD ( $= M_M/M_D^{0.5}$ ). The RMD is similar to the sodium adsorption ratio (SAR) commonly used in the soil science literature. However, Kolstad et al. (2004) noted that the RMD is slightly different from SAR in that: (1) RMD characterizes the permeant solution introduced to the soil, whereas SAR generally describes pore water equilibrated with the soil (although SAR has been used in some cases to describe solutions); (2) RMD is defined in terms of molar concentrations, whereas SAR is defined in terms of normality (e.g., McBride 1994); (3) RMD includes all monovalent and divalent cations in solution, whereas SAR is limited to  $Na^+$ ,  $Ca^{2+}$ , and  $Mg^{2+}$ ; and (4) RMD does not include a factor of 2 in the denominator like SAR. Based on these considerations, the use of RMD may be more appropriate than SAR for evaluating the compatibility of soil-based hydraulic barriers used to contain liquids with solution chemistries that are more complex than those based solely on the concentrations of  $Na^+$ ,  $Ca^{2+}$ , and  $Mg^{2+}$ .

The resulting correlation between the  $k$  of GCLs and the I and RMD of permeant liquids established by Kolstad (2000) and Kolstad et al. (2004) is shown in figure 6. As shown in figure 6, the predicted  $k$  values for the GCL evaluated in this study based on the estimated I of approximately 0.059 M for the simulated animal waste solution (table 2) range from a low of  $6.4 \times 10^{-10}$  cm/s (for RMD > 0) to a high of  $1.1 \times 10^{-8}$  cm/s (for RMD = 0), with a geometric mean of  $2.7 \times 10^{-9}$  cm/s. This range in predicted  $k$  values encompasses all of the measured  $k$  values for the GCL based on permeation with the simulated animal waste solution (table 4). In addition, the geometric mean of  $2.7 \times 10^{-9}$  cm/s for the predicted  $k$  values is very close to the geometric means of the



**Figure 6.** Range in predicted hydraulic conductivity ( $k$ ) values based on the ionic strength ( $I$ ) of the simulated animal waste solution used in this study and the correlation among  $k$ ,  $I$ , and the ratio of monovalent-to-divalent cations (RMD) from Kolstad (2000) and Kolstad et al. (2004).

measured  $k$  values of  $1.4 \times 10^{-9}$  and  $5.5 \times 10^{-9}$  cm/s based on permeation with the simulated animal waste solution under anaerobic and aerobic conditions, respectively. Thus, the values of  $k$  for the GCL measured in this study are consistent with those reported for other GCLs permeated with a variety of liquids (i.e., leachates from municipal solid waste, construction and demolition waste, fly ash, and mine waste). The correlation between the  $I$  and RMD of the permeant liquid and the  $k$  of the GCL proposed by Kolstad (2000) and Kolstad et al. (2004) appears to be equally valid in the present study. Accordingly, actual animal waste solutions with higher ionic strengths than used in this study would be expected to result in higher  $k$  values than those reported in this study.

#### EFFECT OF REDOX POTENTIAL

The redox potential data also may provide some insight as to why the average  $k$  based on permeation of the GCL specimens with the simulated animal waste solution under aerobic conditions is greater than the average  $k$  based on permeation under anaerobic conditions. For example, Shen et al. (1992) permeated specimens of two consolidated clay sediments (i.e., sodium smectites) that were initially in an oxidized state with a solution containing sodium dithionite ( $\text{Na}_2\text{S}_2\text{O}_4$ ), a strong reductant. They found that the  $k$  of the specimens increased by 43% and 92% relative to the  $k$  based on permeating identically prepared specimens with a solution containing sodium sulfate ( $\text{Na}_2\text{SO}_4$ ), a weak oxidant, at similar ionic strength based on  $\text{Na}^+$  (i.e., 115 mM  $\text{Na}^+$ ). This observation, together with previously published information, led Shen et al. (1992) to conclude that the increase in  $k$  was due to the reduction in structural iron (Fe) in the clays from Fe(III) to Fe(II).

In contrast, Kamon et al. (2002) permeated compacted specimens of a Japanese marine clay with solutions containing either  $\text{Na}_2\text{SO}_4$  or  $\text{Na}_2\text{S}_2\text{O}_4$  and found essentially no difference in  $k$ . They attributed the lack of difference in  $k$  to the marine clay existing naturally in a strongly reduced condition ( $E_h = -377$  mV), such that permeation with the

strongly reducing solution containing  $\text{Na}_2\text{S}_2\text{O}_4$  ( $E_h = -500$  mV) had no effect on the structure of the clay.

Although the redox potential of the bentonite in the GCL evaluated in this study was not measured, the higher geometric mean  $k$  for the GCL specimens permeated with the simulated animal waste solution under aerobic conditions relative to the geometric mean  $k$  based on permeation under anaerobic conditions may be due, in part, to somewhat greater reducing conditions existing in the tests performed under aerobic conditions, as reflected by the previously noted lower range of  $E_{h,\text{out}}$  values for these tests (fig. 5). However, additional study is required to confirm this possibility, given the differences between the materials and test procedures used by Kamon et al. (2002) and Shen et al. (1992) versus those used in this study.

#### PRACTICAL IMPLICATIONS

The use of GCLs as alternative liners and/or liner constituents for anaerobic animal waste lagoons will require compliance with the various state regulations governing liners for animal waste containment (table 5). Three of the 12 states listed in table 5 (NM, SD, and TX) regulate the maximum hydraulic conductivity ( $k_{\text{max}}$ ) for the earthen liner exclusive of any consideration of the minimum liner thickness ( $L_{\text{min}}$ ). The  $k_{\text{max}}$  for these three states is  $10^{-7}$  cm/s, except for the possibility in North Carolina where  $k_{\text{max}}$  is  $10^{-6}$  cm/s if bedrock is less than 1.2 m (Parker et al., 1999). Since the  $k$  of the GCL evaluated in this study was substantially lower than either  $10^{-6}$  or  $10^{-7}$  cm/s, the GCL presumably would satisfy the  $k_{\text{max}}$  criterion for earthen liners in these three states, i.e., assuming animal waste with chemical compositions similar to the one used in this study. However, given the potential for different chemical compositions for the animal wastes in these three states, including the possibility for animal wastes containing higher ionic strengths, lower RMD, and/or lower redox potentials, prudence dictates an evaluation of a specific GCL with the specific animal waste prior to regulatory approval. In cases where such an evaluation results in unacceptably high  $k$  values, use of a composite GCL (i.e., standard bentonite-based GCL with a polymer, such as polyethylene or polypropylene, backing on one side) may be required.

Three of the 12 states listed in table 5 (NM, OK, and WI) specify both a  $k_{\text{max}}$  and a  $L_{\text{min}}$  for earthen liners for animal waste lagoons. In all three states,  $k_{\text{max}}$  is  $10^{-7}$  cm/s, whereas  $L_{\text{min}}$  is 30.5 cm (1 ft) in New Mexico, 45.7 cm (1.5 ft) in Oklahoma, and either 45.7 cm for the bottom liner or 61 cm (2 ft) for the liner on the side slopes in Texas. In contrast, the thicknesses of GCLs in an air-dried condition before placement are typically on the order of about 1 cm and can range from about 0.5 to 1.5 cm in thickness after placement, depending on the load placed on the GCL and the extent of swelling of the bentonite (Shackelford et al., 2000). Thus, a GCL would not meet the  $L_{\text{min}}$  criteria for these three states.

A comparison of the results of this study with the regulations for the three states specifying both  $k_{\text{max}}$  and  $L_{\text{min}}$  for earthen liners for animal waste lagoons (i.e., NM, OK, and WI) also can be based on the hydraulic conductance, defined as the hydraulic conductivity divided by the liner thickness. In this case, the maximum values for the hydraulic conductance (i.e.,  $k_{\text{max}}/L_{\text{min}}$ ) range from  $1.6 \times 10^{-9}$  s $^{-1}$  for WI to  $3.3 \times 10^{-9}$  s $^{-1}$  for NM. Based on the results of this study

(table 4) and an assumption of a 1-cm-thick GCL, the GCL would meet the criteria for maximum hydraulic conductance values under an anaerobic condition, but would fail to meet the criteria under an aerobic condition.

However, because of the need to scrape solid manure from the lagoon periodically to maintain the storage volume of the ponds (Reddi and Davalos, 2000; Reddi et al., 2005), protection of the GCL from such scraping likely would be required. One possible method of protecting the GCL would be to place (i.e., without compaction) a sacrificial layer of a cover soil on top of the GCL with a thickness equal to or greater than the  $L_{\min}$ . On the basis that the placed manure would result in bioclogging of the cover soil with a concomitant reduction in the  $k$  of the cover soil (e.g., Miller et al., 1985; Rowsell et al., 1985; Barrington et al., 1987a, 1987b; Barrington and Madramootoo, 1989; Wu et al., 1997; Bonala and Reddi, 2005), this reduction in  $k$  of the cover soil would be separate from any reduction in  $k$  of the GCL due to bioclogging of the GCL. The combination of the GCL and the bioclogged cover soil could effectively result in what is commonly referred to as a composite liner system (e.g., Daniel, 1993). In this case, the overall effectiveness of lagoon containment will be dependent on the composite liner system (GCL plus cover soil) as opposed to just the GCL. However, as noted by Bonala and Reddi (2005), the decrease in magnitude of  $k$  for a soil due to clogging is difficult to estimate. Thus, some assessment of the effect of the  $k$  of the cover soil would be required before the suitability of the composite liner could be determined.

Three of the 12 states listed in table 5 (ID, IA, and NE) regulate a maximum infiltration (seepage) rate ( $I_{\max}$ ) for the earthen liner exclusive of any consideration of  $L_{\min}$ . In addition, three other states (CO, KS, and MO) specify both an  $I_{\max}$  and an  $L_{\min}$  for earthen liners for animal waste lagoons, where the  $I_{\max}$  is different in each of the three states, but  $L_{\min}$  is either 15 cm (KS) or 30.5 cm (CO, MO). In these states, assessment of the suitability of a GCL will require an assessment of the maximum seepage rate through the liner (GCL alone or GCL with cover soil), in accordance with Darcy's law based on the  $k$  of the GCL, the  $k$  of the cover soil, and the hydraulic gradient across the liner (e.g., Reddi et al., 2005).

## CONCLUSIONS

The geometric mean  $k$  of  $1.3 \times 10^{-9}$  cm/s based on permeation of duplicate specimens of the GCL with DIW under aerobic conditions was the same as that based on permeation with DIW under anaerobic conditions. Thus, the condition under which the  $k$  of the GCL was measured based on permeation with DIW had essentially no effect on the measured  $k$ .

All of the values of  $k$  for the GCL measured in this study based on permeation with the simulated animal waste solution were found to be consistent with those reported for other GCLs permeated with a variety of liquids (i.e., leachates from municipal solid waste, construction and demolition waste, fly ash, and mine waste) at the same ionic strength. In addition, the geometric mean  $k$  of duplicate specimens of the GCL permeated under anaerobic conditions with the simulated animal waste solution was  $1.4 \times 10^{-9}$  cm/s, whereas the geometric mean  $k$  of duplicate

specimens of the GCL permeated under aerobic conditions with the simulated solution was  $5.5 \times 10^{-9}$  cm/s. Thus, permeating the GCL with the simulated solution under anaerobic conditions had little effect relative to the  $k$  based on DIW, whereas permeating the GCL with the simulated solution under aerobic conditions resulted in a  $k$  that was 4.2 times higher on average than that based on DIW. Since the ionic strength of the simulated solution is low and the ratio of monovalent to divalent cations (RMD) is high relative to those of other solutions that reportedly have resulted in significant increases (>100 times) in the  $k$  of other GCLs, the higher  $k$  of the GCL tested under aerobic conditions may be due to the lower redox potential (Eh) of the solution under aerobic conditions relative to that under anaerobic conditions. However, these results are incomplete since other factors, such as the biological component of the animal waste stream, were not evaluated in this study. Since animal waste solutions with lower redox potentials and/or higher concentrations of divalent metals (e.g.,  $\text{Ca}^{2+}$ ) than were used in this study may provide different results, additional study in which the  $k$  of GCLs is measured using actual animal waste streams as the permeant liquid is recommended.

For all practical applications where a GCL is used as a substitute for an earthen liner in an animal waste lagoon, a cover soil likely will be required to protect the GCL from periodic scraping of the solid manure from the lagoon to maintain the storage volume of the ponds. In the case where states regulate only a maximum hydraulic conductivity ( $k_{\max}$ ) or maximum infiltration rate ( $I_{\max}$ ), the cover soil merely serves as a sacrificial soil layer that is not counted on to provide any hydraulic resistance such that any type of soil could be used. However, for states that regulate both a  $k_{\max}$  or  $I_{\max}$  and a minimum liner thickness ( $L_{\min}$ ), the cover soil must be considered as part of the liner, because the relative thinness of GCLs will preclude their sole use as a substitute for thicker earthen liners in these states. However, even when the cover soil is to be considered as part of the liner, the hydraulic resistance of the cover soil may not be important, provided that a suitably low  $k$  for the GCL can be maintained. The actual value of  $k$  will depend on the specific criterion for the state with respect to  $k_{\max}$  or  $I_{\max}$  such that evaluating the  $k$  of the GCL being considered using the actual animal waste to be contained will be required. In cases where such an evaluation results in unacceptably high  $k$  values, use of a composite GCL (i.e., polymer-backed GCL) may be required.

The potential use of the GCL evaluated in this study as an alternative to earthen liners used for animal waste lagoons in 12 different states was evaluated. The evaluation showed that the GCL would be suitable for all three states (NC, SD, TX) that require only a  $k_{\max}$  of  $10^{-7}$  cm/s, i.e., assuming animal waste with chemical compositions similar to the one used in this study. However, the use of the GCL for the states regulating  $k_{\max}$  and  $L_{\min}$  (NM, OK, WI),  $I_{\max}$  (ID, IA, NE), or  $I_{\max}$  and  $L_{\min}$  (CO, KS, MO) is more problematic in that the suitability will require an analysis based on Darcy's law that depends on the  $k$  of the GCL, the  $k$  of any cover soil, and the hydraulic gradient across the liner and soil.

## ACKNOWLEDGEMENTS

This project was funded by a grant from the USDA Agricultural Experiment Station at Colorado State University. The GCL evaluated in this study was provided by CETCO Linings Technology Group, Arlington Heights, Illinois. The assistance of Jim Olsta of CETCO is appreciated.

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