

# Compacted artificially cemented soil–acid leachate contaminant interactions: Breakthrough curves and transport parameters

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## Abstract

The transport of contaminants through compacted artificially cemented soil subjected to acid leachate contaminant percolation was analyzed by means of laboratory column tests. The effect of cement content, degree of acidity and hydraulic gradient were evaluated after permeation of several pore volumes of acid leachate contaminant flow through the soil. The pH, electric conductivity and solute breakthrough curves were considered throughout the study. The results showed that the increase of cement content increases the solute pore volumes needed before breakthrough occurred. An increase of the degree of acidity of the percolate and of the hydraulic gradient cause a reduction in the pore volumes needed before breakthrough occurred. The larger the soil cement content, the longer the time required to reach maximum effluent solute concentration. The hydraulic conductivity slightly increased due to cement addition and reduced with increasing degree of acidity of the percolate. Finally, it is possible to state that cement addition to the soil was responsible for increasing retardation coefficient ( $R$ ) and distribution coefficient ( $k_d$ ) values, meaning that the artificially cemented soils have higher capability to retard the propagation of the contamination and amplified affinity with dissolved acid contaminant.

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**Keywords:** Pyritic wastes; Liners; Acidic waters; Coal mining; Artificially cemented soils

## 1. Introduction

Not only in Brazil, but in the entire world, the mining process can generate environmental impacts to water, air and subsurface. Mining can be one of the more important sectors of an economy by stimulating the formation of wealth and society development; however, environmentally responsible operations will limit long-term liability.

As is common in many mines, environmental monitoring of and mining operation impacts within, carboniferous basins are necessary for mineral coal mines in Brazil. In Brazil, coal exploration reached its peak between 1980 and 1990. Brazil-

ian coal reserves reach 32 billions of tonnes, of which 87% is located in Southern Brazil. Although coal mining is an important energy resource, the extraction and use of mineral coal are potentially polluting activities that impact the environment around the mines. Thus, it is critical that extraction activities operate with social responsibility [1]. The environmental impacts of coal exploration are mainly related to soil and water pollution. One major source of pollution is related to the waste derived from coal mining. Commonly, the waste rock contains pyrite ( $\text{FeS}_2$ ) that can react with oxygen and water to produce sulphuric acid solutions ( $\text{H}_2\text{SO}_4$ ).

Shale waste rock containing pyrite has been deposited for decades next to many Brazilian coal mines. The acid mine drainage derived from this waste can be rusty in color, low pH, and have elevated metals concentrations. The acid mine drainage is an environmental liability that has caused damages to the hydraulic resources of the region [2].

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### Nomenclature

$C$	effluent concentration of solute
$C_o$	concentration of solute in the source
$D_{hd}$	hydrodynamic dispersion
$D^*$	effective molecular diffusion
$d$	average diameter of soil particles
$G_s$	density of solids
$k$	hydraulic conductivity
$k_d$	distribution coefficient
$n$	porosity
$Pe$	Peclet number
$R$	retardation
$t$	time
$V$	volume of liquid percolated
$V_o$	volume of voids of specimen
$v_x$	average linear groundwater velocity
$x$	length
$\alpha$	dispersivity coefficient
$\rho_b$	porous media bulk density
$\gamma_d$	dry unit weight of soil mass
$\omega_{optimum}$	optimum moisture content

In Southern Brazil, deposits of coal mine have high amounts of noncombustible mineral substances associated with the coal. During the process, 30–60% of the mining materials are rejected resulting in the production of large amounts of waste that is high in carbon and mineral (including pyrite) materials but has no commercial value. As a result, the waste is stockpiled around the mine. Fig. 1 depicts the stockpiled waste rock adjacent to a stream that discharges to a large river downstream.

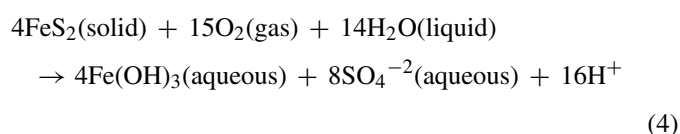
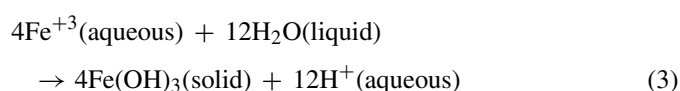
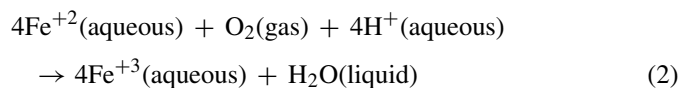
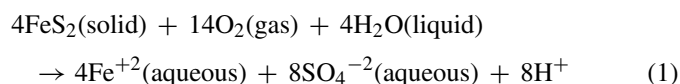
The production of sulphuric acid due to oxidation in the waste rock and subsequent lower pH drainage waters, can mobilize metals within the stockpiled rock and also within the foundation soils, such as iron, manganese, calcium, magnesium and sodium. The pH of the percolate from pyrite wastes ranges from 0.8 to 4.0 [3]. During the summer, high evaporation rates results in



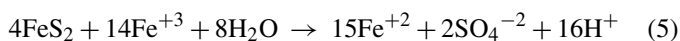
Fig. 1. Mining residues stockpiles from coal extraction.

more concentrated drainage water and the pH can decrease to around 1.2. The pyrite contains about 46% of iron and more than 50% of sulphur dioxide; in the 4% remaining, materials such as arsenic and nickel can be found [4].

Stockpiled pyrite wastes have limited environmental impact if there is no water (e.g., humidity) and/or oxygen available for oxidation to occur [5]. The process of pyrite oxidation occurs according to the Eqs. (1)–(5)



$\text{Fe}^{+3}$  hydrolysis reaction generally occurs in the initial stages of pyrite oxidation limiting the activity of free  $\text{Fe}^{+3}$  in the solution. However, as acidity accumulates in the environment and consequently pH decreases to values smaller than 3.5, hydrolysis reaction is limited, increasing significantly the concentration of  $\text{Fe}^{+3}$  in the solution. In such conditions,  $\text{Fe}^{+3}$  will act as an electron acceptor, becoming the main mechanism of sulphur oxidation, as shown in Eq. (5).



When the pH of the environment is sufficiently acidic,  $\text{Fe}^{+3}$  turns to be the main oxidizer of pyrite and reducing to  $\text{Fe}^{+2}$ . In acid conditions corresponding to pH values smaller than 3, pyrite oxidation due to  $\text{Fe}^{+3}$  occurs 100 times faster than due to  $\text{O}_2$  [6]. The process becomes capable of generating huge amounts of  $\text{H}^+$ .

One approach used to limit oxidation of pyrite waste rock is to store the material in a landfill with bottom and top liners engineered to limit the entrance of oxygen and water from contacting the waste rock. The liners also serve the purpose to limit the transport of acidic waters and contaminant out of the landfill. However, acidic waters in contact with the soil-based liners could result in unexpected reactions that augment the geotechnical and contaminant transport properties of the liner from the as-placed characteristics. Liner integrity could be compromised if the change in soil properties is significant and result in uncontrolled transport of contaminant through the liner. Thus, it is critical to examine the behavior of soil-based liners exposed to acidic waters for examination of reactions that may impact liner performance, and also to investigate and understand the transport of contaminants through the liner.

The first objective of this research is to investigate the influence of acidic waters on the parameters of contaminant transport (e.g., diffusion and advection) through a residual soil under

the variation of the following variables: concentration of sulphuric acid dissolved in percolate, hydraulic gradient and soil cement content. The second objective is to assess the retardation capacity of the soil, variation of hydraulic conductivity during percolation, and any changes to the soil Atterberg limits due to percolation. The benefits of cement addition in the soil in order to decrease impact caused by percolation of low pH waters will be assessed.

## 2. Basic concepts

Advection is the transport of a dissolved solute that occurs due to the flux of water through a porous medium (i.e., soil). For 1D conditions, the change in concentration of a solute with time as a result of advective transport is characterized according to the Eqs. (6) and (7) [7].

$$\frac{\partial C}{\partial t} = -v_x \times \frac{\partial C}{\partial x} \quad (6)$$

where

$$v_x = \frac{k}{n} \times i \quad (7)$$

$C$  = concentration of solute ( $M/L^3$ );  $t$  = time (T);  $v_x$  = average linear groundwater velocity (L/T);  $i$  = hydraulic gradient;  $n$  = porosity;  $k$  = hydraulic conductivity (L/T); and  $x$  = length of interest (L).

In addition to advection, the transport of a solute through a porous media can be influenced by effective molecular diffusion and dispersion. Diffusion, a chemical process, is the transport of a solute from an area of high concentration to low concentration. Dispersion is a mechanical mixing process that results in a solute to move faster than, and slower than, the average linear groundwater velocity. Despite molecular diffusion and hydrodynamic dispersion being distinct mixing processes that influence solute transport, they are mathematically additive and can be combined to form the hydrodynamic dispersion ( $D_{hd}$ ) coefficient, according to Eq. (8).

$$D_{hd} = D^* + \alpha \times v_x \quad (8)$$

where:  $D_{hd}$  = hydrodynamic dispersion ( $L^2/T$ );  $D^*$  = effective molecular diffusion ( $L^2/T$ );  $\alpha$  = dispersivity coefficient (L);  $v_x$  = average linear groundwater velocity (L/T).

For 1D conditions, the change in concentration of a solute with time as a result of hydrodynamic dispersion is characterized according to Eq. (9) [8]:

$$\frac{\partial C}{\partial t} = - \left( v_x \times \frac{\partial C}{\partial x} \right) + D_{hd} \times \frac{\partial^2 C}{\partial x^2} \quad (9)$$

Examination of Eq. (8) reveals that if the effective diffusion coefficient is large in comparison to mechanical dispersion (product of the groundwater velocity and dispersivity), then the hydrodynamic dispersion coefficient is dominated by diffusive processes not dispersive processes. Conversely, dispersive processes dominate the hydrodynamic dispersion coefficient if the dispersive term is large in comparison to the effective diffusion coefficient. It is recognized that examination of the Peclet

number ( $Pe$ ) can be used to assess if diffusion or dispersive processes are significant [7]. Generally, a diffusion dominated system occurs for  $Pe$  numbers less than about  $10^{-1}$ , dispersive dominated system for  $Pe$  numbers greater than about  $10^{-1}$ . Peclet number is characterized according to the Eq. (10):

$$Pe = \frac{d \times v_x}{D^*} \quad (10)$$

where  $Pe$  = Peclet number;  $d$  = average diameter of soil particles (L).

The partition (or distribution) coefficient ( $k_d$ ) is very important in estimating the potential for the adsorption of dissolved contaminants in contact with soil. As typically used in fate and contaminant transport calculations,  $k_d$  is defined as the ratio of the contaminant concentration associated with the solid ( $C_{solid}$ ) to the contaminant concentration in the surrounding aqueous solution ( $C_{solution}$ ) when the system is at equilibrium, as showed in Eq. (11) [9].

$$k_d = \frac{C_{solid}}{C_{solution}} \quad (11)$$

where  $k_d$  = distribution coefficient ( $L^3/M$ );  $C$  = concentration ( $M/L^3$ ).

Retardation factor ( $R$ ) is a dimensionless parameter characterizing the retarding effect of adsorption on solute transport. A higher value of  $R$  means that the soil is able to retard the breakthrough of the contamination more efficiently. In this work, the retardation coefficients were estimated directly from the breakthrough curve [10]. From a known value of retardation factor, it is possible to determine the distribution coefficient, according to the Eq. (12) [11], where  $v_x$  is the groundwater velocity,  $v_s$  is the solute velocity,  $\rho_b$  is the bulk density of the soil,  $k_d$  is the distribution coefficient and  $n$  is the porosity of the media.

$$R = \frac{v_x}{v_s} = 1 + \frac{\rho_b \times k_d}{n} \quad (12)$$

## 3. Experimental program

### 3.1. Experimental conditions

Well-controlled column tests were completed on compacted soil and soil–cement mixtures permeated with neutral to acidic waters to analyze the rate of solute transport. Three different soil–cement mixtures were tested: 0, 1, and 2% on a dry mass basis. The soil pH with no Portland cement added was 4.7. The addition of 1 and 2% cement increased the pH of soil–cement mixture to a range between 9 and 12. Each of the compacted soil types were permeated with distilled water that contained either 0, 2, or 6% sulphuric acid under a hydraulic gradient of 10 and 20. Sulphuric acid was employed to decrease the percolate pH to represent the drainage waters emanating from oxidized pyrite waste rock. In total, eighteen different column tests were completed to isolate how acidity of the percolate water and cement additive influence contaminant transport through the compacted soil. The experimental program is provided in Table 1.

Table 1  
Experimental program

Cement added (%)	Sulphuric acid in the solution (%)	Hydraulic gradient	Bulk density ( $\text{kN/m}^3$ )	Porosity	Total of tests carried out
0; 1; 2	0; 2; 6	10; 20	17.6	0.34	18

### 3.2. Soil properties

The soil used for the experiments is a residual soil from Botucatu formation (BRS), sampled in a natural deposit in the metropolitan region of Porto Alegre city, Southern Brazil. This soil is relatively well characterized as a result of recently completed studies [12–14].

The soil without cement has liquid limit of 23.8%, plasticity limit = 13% [15] and density of solids ( $G_s$ ) = 26.5  $\text{kN/m}^3$  [16]. Any changes in these soil index properties will be quantified by analyzing the Atterberg limits after the soil without cement additive is permeated with the percolate acid solution.

Each soil–cement mixture was compacted to the maximum standard Proctor [17] dry unit weight of soil mass ( $\gamma_d = 17.6 \text{ kN/m}^3$ ) and optimum water content ( $w_{\text{optimum}} = 14.2\%$ ).

The material contains approximately 5.0% clay, 36.4% silt, and 58.6% sand (52.6% fine sand and 6.0% medium sand). Thus, the soil is classified as silty sand (SM) according to Unified Classification System [18].

### 3.3. Equipment

A stainless steel column, which contains a 7 cm diameter and 10 cm long soil sample, was manufactured for this study (see Fig. 2). The column is equipped with a top cap that allows liquid to enter the sample, as well as, a piston entrance to transmit a load to the sample, if needed. The column was installed in

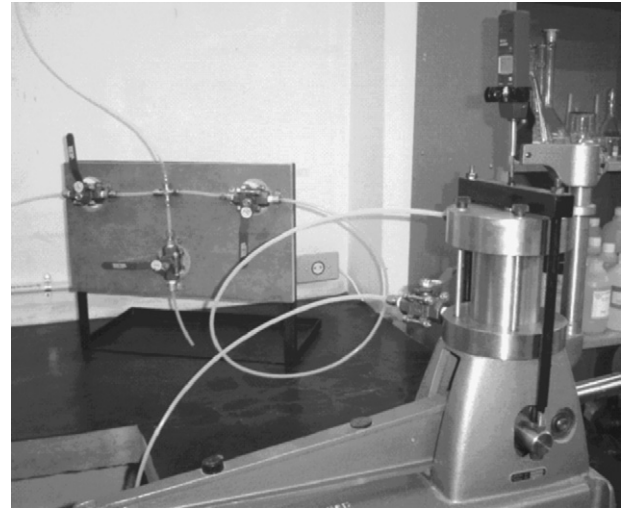


Fig. 3. Modified odometer test.

conventional consolidation equipment (see Fig. 3). An effluent exit was located at the base of the sample. Filters were used at the bottom and on the top of the specimens to avoid the loss of soil out of the equipment.

### 3.4. Methods

After the soil/soil–cement was compacted into the column, the specimens were submitted to the percolation of distilled water until the electric conductivity and volume of water entering and exiting the column were similar. Electrical conductivity provides a measure of the ionic concentration and was used as a surrogate for a directly measured dissolved solute. After the specimen was stabilized to distilled water, it was then submitted to acid percolation. The influent and effluent fluid samples were tested for electrical conductivity and pH with time using standard electrical probes (measuring conductivity in increments of 1.0  $\mu\text{S/cm}$ ) connected to an analogical/digital converter. These electrical probes allow measuring also the pH and temperature of the liquid. Measurements of the volume of solution entering and effluent exiting the specimens were also carried out.

## 4. Results and discussions

### 4.1. Transport of contaminants

The impact of acid solution percolation was visually detected by plotting the relative concentration of solute to the cumulative pore volumes ( $V/V_0$ ) passed through the column. The breakthrough curves obtained from column tests are presented in Fig. 4 (specimens with 0% of cement), Fig. 5 (specimens with 1% of cement) and finally Fig. 6 (specimens with 2% of cement).

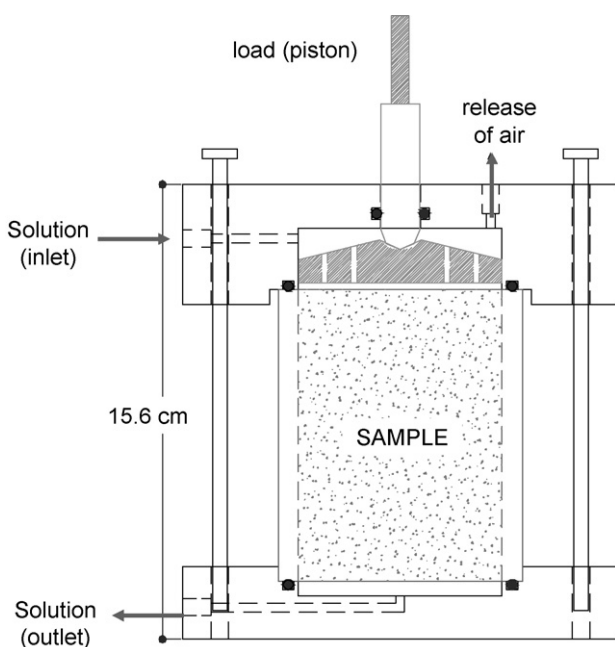


Fig. 2. Modified odometer test project.

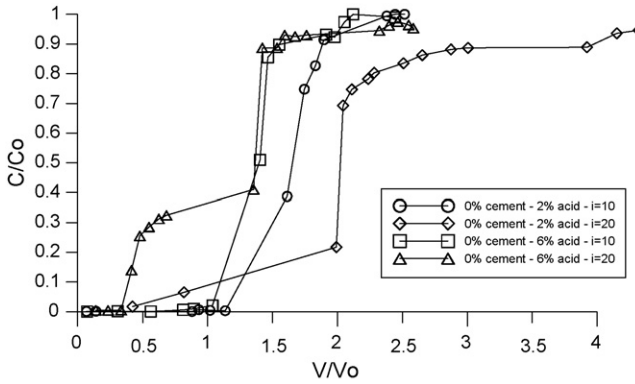


Fig. 4. Breakthrough curves, 2 and 6% of sulphuric acid in the solution—0% of cement added.

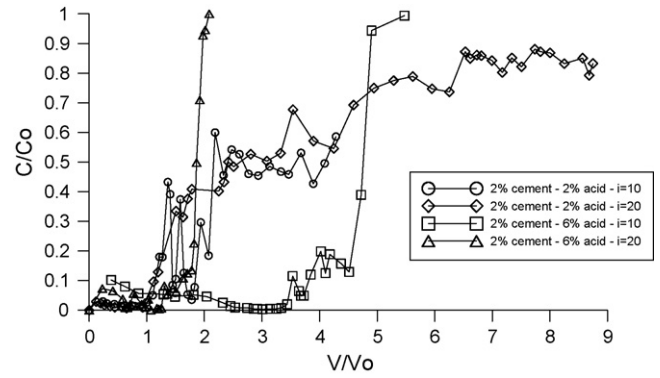


Fig. 6. Breakthrough curves, 2 and 6% of sulphuric acid in the solution—2% of cement added.

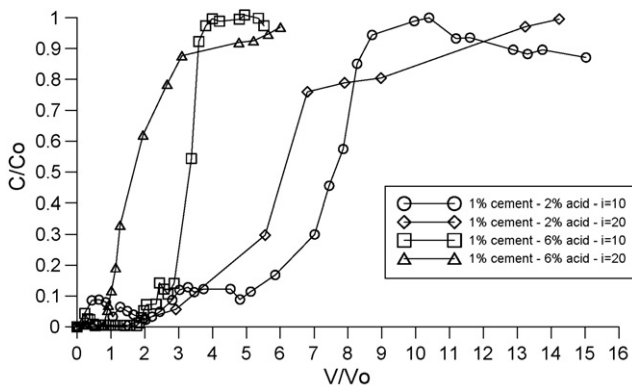


Fig. 5. Breakthrough curves, 2 and 6% of sulphuric acid in the solution—1% of cement added.

The relative concentration ( $C/C_0$ ) was determined based on conductivity values ( $C/C_0 = \text{conductivity of effluent}/\text{conductivity of influent}$ ) on the conductivity meter. The use of electrical conductivity for detecting the breakthrough of the contamination by acid

waters was considered very appropriate, according to calibration tests completed previously of the experimental program.

In Fig. 4 (specimens with 0% of cement) was observed that increasing the hydraulic gradient from 10 to 20 results in solute breakthrough before  $V/V_0$  reaches unity, which means that the solute reached the bottom of the sample prior to one pore volume of acid solution percolated through the soil column. The behavior is more pronounced for the 6% sulphuric acid solution compared to the 2% sulphuric acid solution, and is suggestive of preferential pathways for fluid flux through the sample.

As the ground water velocity is low (according Table 2) and the Peclet number is in the range of 0.1–1, it is assumed that hydrodynamic dispersion is totally dependent on the diffusion, and the second term of the Eq. (8) (advection dependent) can be disregarded. Thus, assuming that hydrodynamic dispersion is equal to the diffusion ( $D_{hd} = D^*$ ), and it is not dependent on the flux, and only the advection and diffusion are responsible for the transport of the contaminants through the liner.

The grain size of the soil sample percolated by a solution with 6% of sulphuric acid was measured for the top (influent

Table 2  
Coefficients of contaminant transport—global result

Cement added (%)	Sulphuric acid in the solution (%)	Hydraulic gradient ( <i>i</i> )	Retardation ( <i>R</i> )	Peclet ( <i>Pe</i> )	$D_{hd}$ (cm <sup>2</sup> /s)	$k_d$ (cm <sup>3</sup> /g)	<i>k</i> (cm/s)	$v_x$ (cm/s)	Total time of testing (h)	
0	0	10	–	–	–	–	$2.05 \times 10^{-5}$	$6.21 \times 10^{-4}$	24	
		20	–	–	–	–	$2.31 \times 10^{-5}$	$7.00 \times 10^{-4}$	24	
	2	10	1.64	1.481	$9.61 \times 10^{-4}$	0.0800	$4.45 \times 10^{-6}$	$9.79 \times 10^{-5}$	85.9	
		20	1.87	1.239	$1.35 \times 10^{-3}$	0.1077	$3.11 \times 10^{-6}$	$1.89 \times 10^{-4}$	124.2	
6	10	10	1.57	0.551	$8.98 \times 10^{-4}$	0.0712	$1.95 \times 10^{-6}$	$5.92 \times 10^{-5}$	119.4	
		20	1.13	0.242	$4.91 \times 10^{-3}$	0.0157	$1.89 \times 10^{-6}$	$1.14 \times 10^{-4}$	79.4	
	1	0	10	–	–	–	–	$2.65 \times 10^{-5}$	$8.03 \times 10^{-4}$	120
			20	–	–	–	–	$2.76 \times 10^{-5}$	$8.36 \times 10^{-4}$	120
2		10	7.04	0.269	$1.54 \times 10^{-3}$	0.7594	$4.33 \times 10^{-6}$	$4.14 \times 10^{-5}$	1055	
		20	6.69	1.119	$1.11 \times 10^{-3}$	0.7156	$2.40 \times 10^{-6}$	$1.24 \times 10^{-4}$	971	
6	10	3.18	0.303	$3.84 \times 10^{-4}$	0.2740	$3.83 \times 10^{-7}$	$1.16 \times 10^{-5}$	1100		
	20	2.13	1.727	$1.34 \times 10^{-4}$	0.1420	$3.90 \times 10^{-7}$	$2.37 \times 10^{-5}$	814		
2	0	10	–	–	–	–	$4.11 \times 10^{-5}$	$1.25 \times 10^{-3}$	120	
		20	–	–	–	–	$4.00 \times 10^{-5}$	$1.21 \times 10^{-3}$	120	
	2	10	5.00	0.022	$7.71 \times 10^{-4}$	0.5030	$5.62 \times 10^{-8}$	$1.71 \times 10^{-6}$	7892.5	
		20	3.86	0.250	$2.39 \times 10^{-4}$	0.3590	$1.01 \times 10^{-7}$	$5.98 \times 10^{-6}$	4085.6	
	6	10	4.41	0.135	$6.46 \times 10^{-4}$	0.4290	$2.88 \times 10^{-7}$	$8.75 \times 10^{-6}$	2264	
		20	1.76	0.214	$1.91 \times 10^{-4}$	0.0950	$6.72 \times 10^{-8}$	$4.09 \times 10^{-6}$	2218	

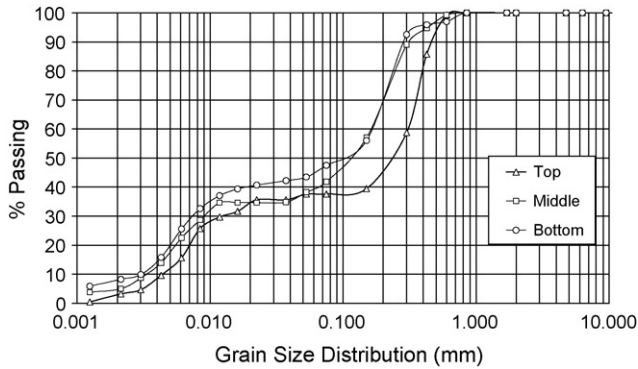


Fig. 7. Grain size distribution test—sample (0% of cement) percolated by a solution with 6% of sulphuric acid.

end), middle, and bottom (effluent end) sections of the sample, and shown in the Fig. 7. The grain size results measure a greater percentage of fine particles towards the effluent end of the soil sample and indicates that fine particles of the soil are migrating to the bottom of the sample, making the top more porous which may give rise to preferential pathways for fluid flux compared to the initial conditions. The breakthrough curves suggest that higher acid content in the percolate, as well as, larger hydraulic gradients result in greater rearrangement of the soil compared to initial conditions. The mechanisms for this behavior may be due to a combination of deflocculation or dissolution of soil matrix and mobilization and entrainment of soil particles spatially within the soil column, resulting in fine particles migrating more intensively to the bottom of the sample and the solute transporting through the soil column faster under gradient 20 than under gradient 10.

Comparing similar cases without cement additive (Fig. 4) to 1% of cement additive (Fig. 5), the case with cement additive required a larger number of pore volumes to percolate through the column before solute breakthrough occurred. Under low concentration of acid in the solution (2%), the addition of 1% cement additive retarded solute breakthrough to a greater extent than the 6% sulphuric acid solution. Additionally, the 2% sulphuric acid solution case compared to the 6% sulphuric acid case required a larger number of pore volumes to pass through the soil before a maximum effluent solute concentration was reached. Regardless of sulphuric acid content in the percolate, soil samples with 1% cement (Fig. 5) required a greater number of pore volumes to reach maximum effluent solute concentration compared to soil samples with no cement content (Fig. 4).

The addition of 2% of cement, when compared with similar cases without cement additive, also increased the pore volumes required before solute breakthrough, mainly under high concentrations of sulphuric acid in the solution (Fig. 6). Under low acid concentrations, the behavior of the breakthrough curves was not regular; the curves show a variation during the rise which may be attributed to the high buffer capacity of the soil matrix relative to the low acid content of the percolate and the resulting attenuation. The tests carried out under low concentration of acid (2%) and 2% of cement additive were terminated before the relative concentration reached unity once that the equipment needed maintenance. The duration of this testing was more than

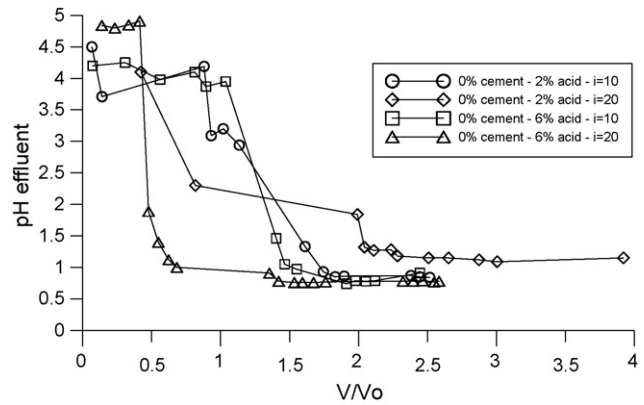


Fig. 8. pH of the percolate versus volume percolated (0% of cement added).

7800 h (more than 10 months) and the breakthrough curves did not reach a value of unity, indicating that the buffering capacity of the soil matrix was sufficient to achieve a residual amount of attenuation.

The variation of the pH (outlet) versus cumulative pore volume for samples with 0% of cement added, under hydraulic gradient 10 and 20 is provided in Fig. 8. The effluent pH decreases from an initial acidic pH of about 4.7 at a similar cumulative pore volume at which the relative concentration of contaminants ( $C/C_0$  outlet) increases (see Fig. 4), corroborating the breakthrough curves trends.

Fig. 9 shows the variation of the pH (effluent) versus cumulative pore volume for samples with 1% of cement added, percolated under hydraulic gradient 10 and 20. Due to insertion of 1% cement, the pH of the soil before acidic permeation increased to about 12 compared to a pH value of about 4.7 if no cement additive. The higher initial pH of the pore waters in cemented samples is a reflection of the greater buffering capacity of the soil matrix to acid percolates.

Regarding time of testing, the amount of cement added in the samples was the most important variable in controlling the breakthrough time of solute through the soil and is largely a reflection of the lower hydraulic conductivity of soil–cement mixtures (see Table 2). As shown in Fig. 10, the greater the cement content of the soil the longer the time for relative concentration of solute in the percolate to approach unity.

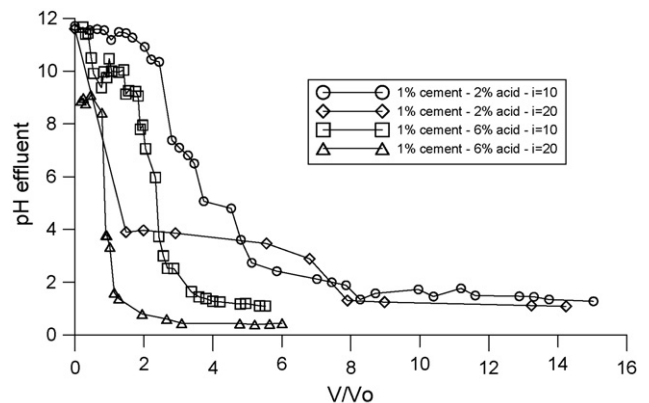


Fig. 9. pH of the percolate versus volume percolated (1% of cement added).

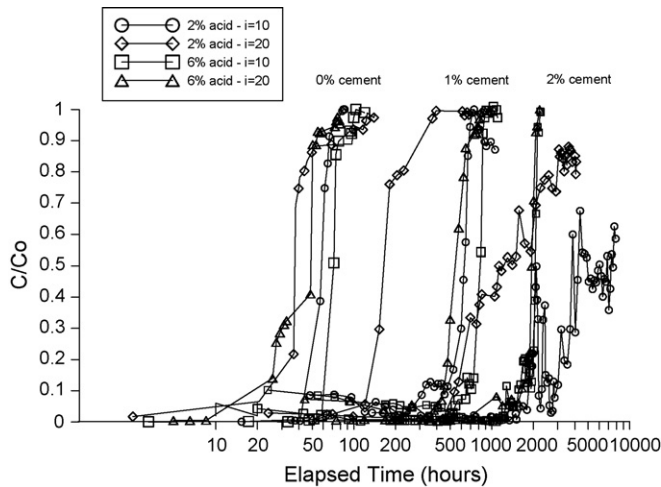


Fig. 10. Relative concentration of sulphuric acid (outlet) versus total elapsed time of testing.

The results of solute transport coefficients for the various cases analyzed are summarized in Table 2. The retardation coefficient ( $R$ ) ranged from 1.13 to 1.87 for uncemented samples with higher retardation coefficients for a lower sulphuric acid concentration in the percolate. The addition of 1% of cement increased the retardation coefficient for all combinations of hydraulic gradients and concentrations of acid in the percolate, with values ranging from 2.13 to 7.04. Additions of 2% of cement did not result in higher retardation coefficients compared to the 1% cement case.

The hydrodynamic dispersion values ranged from about  $10^{-4}$   $\text{cm}^2/\text{s}$  to approximately  $10^{-3}$   $\text{cm}^2/\text{s}$ . In uncemented samples, an increase of the hydrodynamic dispersion was deduced when the hydraulic gradient increased from 10 to 20 (from  $10^{-4}$  to  $10^{-3}$   $\text{cm}^2/\text{s}$ ); conversely, cemented samples had a different behavior, where increases in the hydraulic gradient caused a decrease of the hydrodynamic dispersion, more accentuated for samples with 1% of cement. Samples with 2% of cement addition had changes only in cases where the hydraulic gradient changed from 10 to 20, for increases of the concentration of acid in the solution did not cause changes in the hydrodynamic dispersion.

The lowest values of the distribution coefficient ( $k_d$ ) were observed from samples percolated by solutions with 6% of sulphuric acid dissolved, where as, the greater the cement content the higher  $k_d$  values. As  $k_d$  is the measure of the affinity between the soil matrix and the contaminant, the greater the cement content in the soil, the greater the acid-buffering capacity of the soil matrix.

The hydraulic conductivity ( $k$ ) of samples percolated by acid waters changed according to the amount of cement added and amount of acid dissolved in the solution. The larger changes observed occurred when the concentration of acid in the solution changed from 0 to 2%. These changes were more expressively seen in samples with 2% of cement added, where  $k$  changed from about  $10^{-5}$   $\text{cm/s}$  (samples percolated by distilled water only) to around  $10^{-8}$   $\text{cm/s}$  (samples percolated by solutions with 6% of acid).

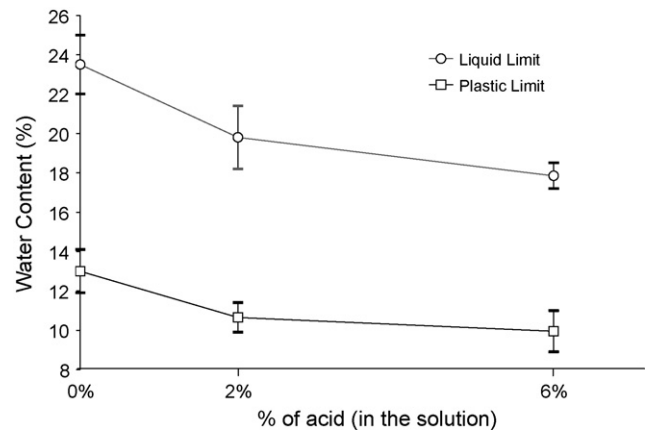


Fig. 11. Atterberg limits variation due to low-pH percolations— $i = 10$ .

#### 4.2. Atterberg limits

Fig. 11 presents the average Atterberg limits (as well as the Standard Deviation observed in the tests carried out), for specimens with no cement addition and percolated by solutions with 0, 2 and 6% of sulphuric acid dissolved in the water. The specimens tested were sampled after the percolation tests ( $C/C_0 = 1.0$ ), and immediately submitted to the Atterberg limit tests. The liquid limit was the most affected by acid percolations which reduced from 23.8% (sample percolated only by distilled water) to approximately 18% after percolation with a 6% sulphuric acid solution. The plasticity limit also changed due to acid percolation, from about 13% (sample without contamination) to around 10%, after percolated by a solution with 6% sulphuric acid solution. During the tests, it was possible to detect a loss of workability of the material after percolated by acid solutions. The loss of workability of the material was possibly caused by the alteration of clay fraction elements (smaller particles) of the soil due to attack of sulphuric acid.

#### 5. Conclusions

The effect of adding cement to a soil subjected to acid mine drainage percolation (considering distinct degrees of acidity and hydraulic gradient) was studied by means of laboratory column tests. The main conclusions of such study are as follows:

- The greater the amount of cement added to the soil, the larger the needed time of testing to reach the maximum ratio of effluent concentration to influent concentration ( $C/C_0$ ) in the percolate.
- An increase in degree of acidity of the solute and of the hydraulic gradient caused a reduction in the pore volumes needed to initiate solute breakthrough.
- The results showed that the increase of cement content increased the solute pore volumes needed to initiate solute breakthrough.
- The hydraulic conductivity slightly increases due to cement addition and reduces with increasing degree of percolate acidity.

- Cement addition to the soil was responsible for increasing the retardation coefficient ( $R$ ) and distribution coefficient ( $k_d$ ) values compared to non-cemented soils, meaning that the artificially cemented soil has higher capability to delay the propagation of the contamination and amplified affinity with dissolved acid contaminant, thereby increasing the acid-buffering capacities.
- The Atterberg limits of uncemented specimens were affected by the acidity of the percolate, which may result in a loss of workability of the material after percolated by acid solutions.
- The design implications of these findings are that adding cement to soils increases the soil matrix pH thereby increasing the acid-buffering capacity when permeated with acidic waters and hence amplifying the capacity to attenuate pollutants in acid leachate contaminants.

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