Impact of biological clogging on the barrier performance of landfill liners[1]

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[1] This is an Accepted Manuscript of an article published by *Journal of Environmental Management*. The final publication is available online via https://doi.org/10.1016/j.jenvman.2018.05.039

Abstract

The durability of landfill mainly relies on the anti-seepage characteristic of liner system. The accumulation of microbial biomass is effective in reducing the hydraulic conductivity of soils. This study aimed at evaluating the impact of the microorganism on the barrier performance of landfill liners. According to the results, Escherichia coli. produced huge amounts of extracellular polymeric substances and coalesced to form a confluent plugging biofilm. This microorganism eventually resulted in the decrease of soil permeability by 81% - 95%. Meanwhile, the increase of surface roughness inside the internal pores improved the adhesion between microorganism colonization and particle surface. Subsequently, an extensive parametric sensitivity analysis was undertaken for evaluating the contaminant transport in landfill liners. Decreasing the hydraulic conductivity from 1×10^{-8} m/s to 1×10^{-10} m/s resulted in the increase of the breakthrough time by 345.2%. This indicates that a low hydraulic conductivity was essential for the liner systems to achieve desirable barrier performance.

Keywords:

Landfill liner; biological clogging; hydraulic conductivity; barrier performance.

1. Introduction

The production of municipal solid waste (MSW) significantly increased with the rapid growth of population over the past two decades. The associated question is how to manage these solid wastes (Tang et al., 2017, 2018). General management methods towards MSWs include recycling, composting, anaerobic digestion, incineration, dumping into the sea and landfilling (Tang et al., 2015). Due to the low cost, sanitary landfilling is the most prevalent method to eliminate MSWs especially in developing and underdeveloped countries (Zhan et al., 2014). For instance, in South Africa and China, approximately 100% and 73% of collected MSWs were landfilled (Blight, 2006; NBSC, 2013).

Although landfill management is widely adopted, it becomes an emerging threat to the environment as well. The landfill usually results in the groundwater contamination due to leachate leakage from landfill sites (Sibiya et al. 2017). Therefore, the anti-seepage and contaminant sorption capability of liners is crucial for landfill design. Rajasekaran et al. (2005) pointed out that the transport of contaminants through liner systems mainly depends on the permeability of bottom liner. To prevent leachate from leaking out of landfill, the standard for MSW sanitary landfill (CJJ 176-2012, China) requires that a compacted clay liner should have a minimum depth of 2 m, and have a maximum hydraulic conductivity of 1×10^{-7} cm/s. Composite liners are extremely popular in modern landfills, which usually consist of either a geomembrane (GMB) and a compacted clay liner (CCL), or a GMB, a geosynthetic clay liner (GCL), and a soil liner (SL) (Rowe et al., 2004; Xie et al., 2018). Parastar et al. (2017) concluded that the natural clay soil was a preferred liner material because of its high sorption capacity, long-term structural stability, and low permeability. However, clay minerals are natural non-renewable resources, but construction of a landfill liner always consumes a large amount of clay minerals to meet the anti-seepage requirements and contaminant blocking capability (Wu et al., 2017). Furthermore, in China, there are more mountainous areas than plain areas (Li et al., 2017). This results in a problem that natural clays suitable for landfill liners may not be locally available. Thus, it is critical to pretreat natural clay to achieve suitable hydraulic conductivity, which can save a large quantity of clay soil during the construction of landfill liner.

Microorganisms can develop biofilms in many natural and engineered porous media systems. Biofilm barriers are structures made by stimulating the activity of microorganisms in soils (Rowe 2005). An excessive growth of bacteria in soil causes the biofilm-induced pore clogging, which thereby provides substantial decrease in permeability and groundwater flow (Rowe and Yu, 2013; Yu and Rowe, 2012a, 2012b). Rowe and Booker (1995) found that leachate passing through the drainage layer of MSW landfills can induce clogging, which raises the growth of biomass. Dennis and Thrner (1998) evaluated the reduction of permeability due to the formation of biofilm, and concluded that the bacterial treatment significantly decreased the hydraulic conductivity by 1-3 orders of magnitude. Kirk (2012) confirmed that biomass greatly remains intact after acidification and continue reducing hydraulic conductivity of porous medium, even when considerable death occurs. Hence, biological clogging is effective in forming grout curtains to reduce the migration of heavy metals and organic pollutants, and thereby preventing leaking construction pit, landfill, or dike. However, the types of microorganisms used in the previous studies were not the main bacteria in the landfill leachate. It is known that a high salinity level and a low oxygen supply of leachate dramatically limit the growth of voluminous microorganisms (Aziz et al., 2013). Grisey et al. (2010) found the presence of microorganisms in leachate. They investigated the seasonal variations in abundance of total coliforms, Enterococci, Salmonella, Pseudomonas aeruginosa, and Staphylococcus aureus, and found that total coliforms were predominant in the landfill leachate. Herein, total coliforms are defined as aerobic or facultatively anaerobic, oxidase negative bacteria (Matejczyk et al., 2011). Organisms of genera such as Citrobacter, Enterobacter, Escherichia, Serratia, Klebsiella are also included (Baudisova, 1997). Among the coliforms, the Escherichia coli population was found extremely high in the leachate (Threedeach et al., 2012).

The traditional liner systems are designed solely dependent on the permeability of material. However, even barriers with a zero permeability cannot completely prohibit the release of contaminants. This is because fluid flow will still occur from molecular diffusion of contaminants across the barrier (Daniel and Shackelford, 1988). Once the leachate releases into the soil, it will interact physically and chemically with both the adjacent ground water and the soil matrix (Yeh and Tripathi, 1989). Numerical modelling is capable of predicting the movement and transfer of contaminants in the landfill liner systems (Boddula and Eldho, 2017). Consequently, the assessment of groundwater pollution usually relies on an accurate numerical model for proper management and remediation of the contaminated sites (Xie et al., 2016). Chen et al. (2015) concluded that the main mechanism for inorganic contaminant transport in the composite liners is advection through GMB defects and advection-dispersion in the underlying GCL, CCL, or attenuation layer. They developed a breakthrough time based design method for landfill composite liners which ignored influences of both diffusion and advection. However, the advective transport of organic compounds through GMB defects cannot be ignored, especially in the case with a high leachate head (Xie et al., 2015).

To address the aforementioned research needs, this study evaluated the influence of biological clogging on the hydraulic properties of liner material based on a modified permeameter. The mechanisms of biological clogging were investigated through a series of physical and chemical experiments (e.g., Nitrogen adsorption (N_2-BET), X-ray Diffraction (XRD), X-ray Fluorescence (XRF), Scanning Electron Microscope (SEM), etc.) Subsequently, the finite element models were developed to investigate the transport of contaminant through a natural soil barrier system. The contaminant breakthrough time was calculated and the factors that affected breakdown time were examined. The final section summarized the major findings of this study.

2. Materials and methods

2.1 Soils

The soils used in the experiment were clay and sand. The clay was locally sampled from Tianzhiyun area in Suzhou, China (N31°32’, E120°56’). To avoid the disturbance of environmental change on soil properties, the clay was obtained from 3 ~ 4 m below the ground surface. The quartz-based sand with a quartz content of 99.3% was collected from Fengyang, Anhui, China. The particle size of the sand was 0.15 ~ 0.075 mm. The clay properties were summarized in Table 1. As shown in Table 1, the natural moisture content of the clay was 23.78%. The liquid limit and plastic limit of the clay were 41.54% and 16.57%, respectively. The swelling index was 2 mL/2g-solid, which demonstrated that the clay used in this experiment contained limited amount of expansive mineral (e.g., montmorillonite). The clay was slightly acidic and had a pH of 6.6, which was attributed to the acid rain in Suzhou region. The measured Electrical Conductance (EC) was 0.05 mS/cm, which represented that the soluble salt content in the clay was low. The particle size distributions of clay and sand were shown in Fig. 1.

For the N_2-BET adsorption tests, the correlation coefficient was 0.999. This indicated that the results obtained were reliable. The specific surface area of the clay used in this study is 24.49 m^2/g. The average pore diameter is 8.23 nm, and the total pore volume is 5.04×10^{-2} cm^3/g. The XRD spectra is shown in Fig. 2a (RAD-2B, Rigaku Corporation, Japan). Note that the experimental clay contained quartz (SiO_2), which was observed at 2θ = 20.85°, 26.58°, 45.76°, 73.42°, 75.61°, 79.83° and 90.83°. XRF results confirmed the presence of SiO_2 and SiO_2 accounts for 58.12% of the total soil (JSX-3400R, JEOL, Japan). Meanwhile, other mineral contents such as Albite, Calcite, Phosphosiderite, Antigorite, Magnesite and Monticellite were also found out at 2θ = 27.95°, (39.42°, 81.40°), (36.48°, 42.42°), (40.24°, 59.90°), 68.10° and 50.07°. The existence of Antigorite was proven by the SEM (SU-8020, Hitachi, Japan) image of clay, which was shown in Fig. 2b.

Table 1 Physical and chemical properties of studied soil

Property	Standard	Unit	Value
Physical-chemical properties			
Natural moisture content	JIS A 1203	%	23.78
Swelling index	ASTM D 5890-06	mL/2g-solid	2
Liquid limit	GB/T 50123-1999	%	41.54
Plastic limit	GB/T 50123-1999	%	16.57
pH	JGS 0211		6.6
EC	JGS 0212	mS/cm	0.05
N_2-BET			
Correlation coefficient (R^2)			0.999
Specific surface area		m^2/g	24.49
Average pore Diameter		nm	8.23
Total pore volume		cm^3/g	5.04×10^{-2}

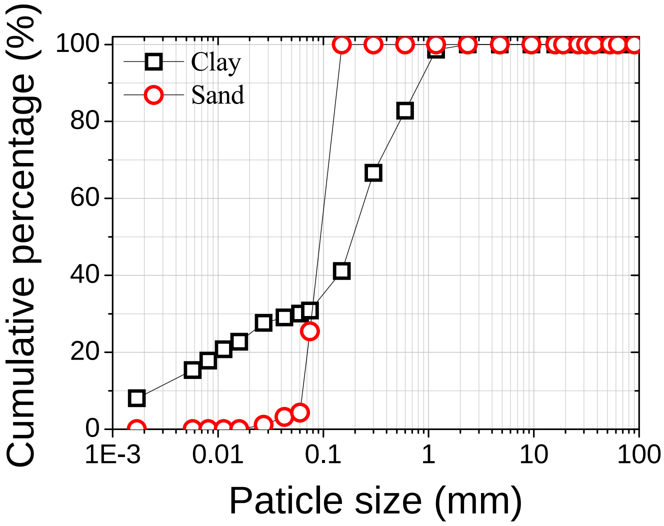
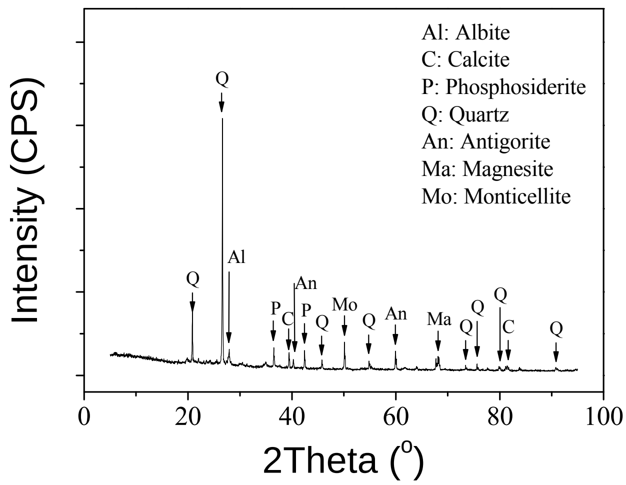
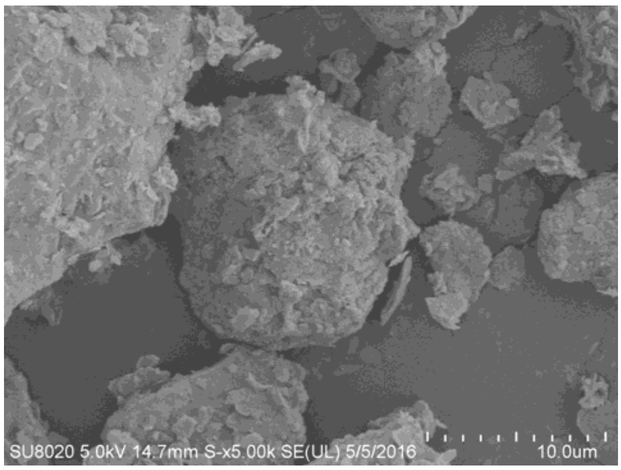


Fig.1 Particle size distribution of clay and sand;





(a) (b)

Fig.2 Micro character of the used soil. (a) XRD spectra of soil; (b) SEM picture of the clay.

2.2 Bacteria used and permeating liquids

The type of bacteria used throughout the study was Escherichia coli, which was the most common bacteria in landfill leachate (Grisey et al. 2010). Escherichia coli was gram-negative, facultative anaerobic, rod-shaped with 0.5 μm in diameter and 1 - 3 μm in length (Ren et al. 2013). This bacterium can be cultured easily and inexpensively in the laboratory, and more importantly, it is non-pathogen (Sugnaux et al., 2013). Escherichia coli has a wide tolerant range for pH (i.e., from 4.5 to 8.0) and for growth temperature (i.e., from 7 to 49.5°C). On the other hand, E. coli can survive in an extreme low oxygen concentration (i.e., a nM level) (Stolper et al., 2010). These characteristics make Escherichia coli an appealing candidate for the environmental application of biofilm. Prior to treatment, the bacteria were stored in centrifuge vials at 4°C.

The two permeant liquids used in the tests were deionized water (DIW) and culture medium (CM). The DIW was used as both the saturated solution and the permeant liquid to obtain initial stable hydraulic conductivity of clay prior inoculating bacteria into the clay. As a permeant liquid, the CM was the Luria–Butani (LB) medium. Solutions of LB medium were prepared by adding the following constituents to DIW to make 1000 mL solution: 10 g tryptone; 5 g yeast extract; 10 g NaCl. The pH was maintained at 7 ± 0.2 through the addition of HCl (0.1 N) or NaOH (0.1 N). The media were sterilized by autoclaving at 0.15 MPa and 121°C for 20 min.

2.3 Experimental program

In this study, a new apparatus was developed to measure the effects of bacteria on the hydraulic conductivity of clay-sand mixtures, which was shown in Fig. 3. Different from the traditional permeameter, a spiral wire and a wire embedded in the top cap and over the cutting ring were used to replace the top porous stone. Meanwhile, a wire loop, a well-shaped wire and a wire were stacked up in the base pedestal to replace the bottom porous stone. Traditionally, filter paper is used between stones and soil to prevent clogging of the stones with soil fines. However, the trial tests revealed that bacteria carried with the pore fluid tended to plug filter paper, which resulted in the measured k value less than the actual k value. Thus, the filter paper was replaced by 0.058 mm pore size nylon gauze in this study. The wide aperture of nylon gauze effectively prevented plugging in the test.

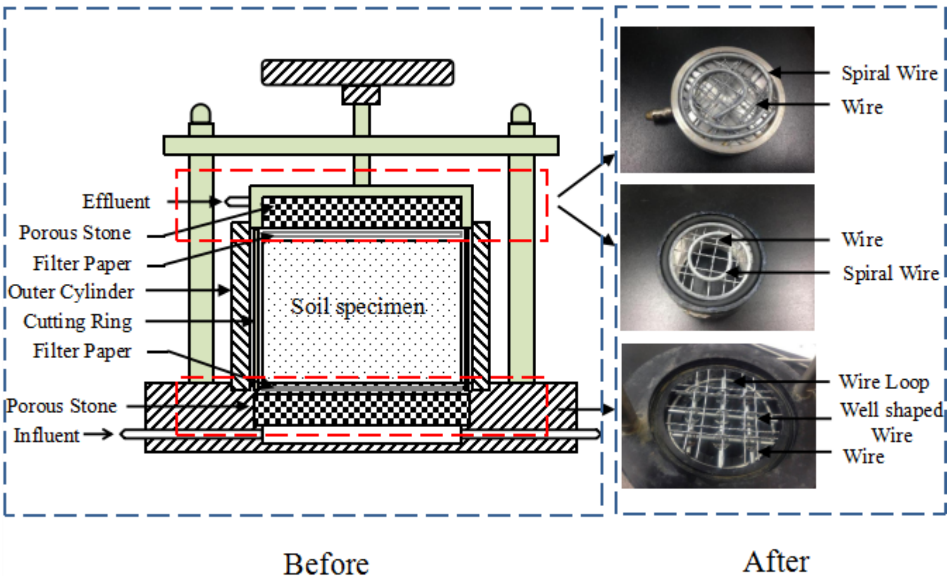


Fig. 3 Instrument improvement

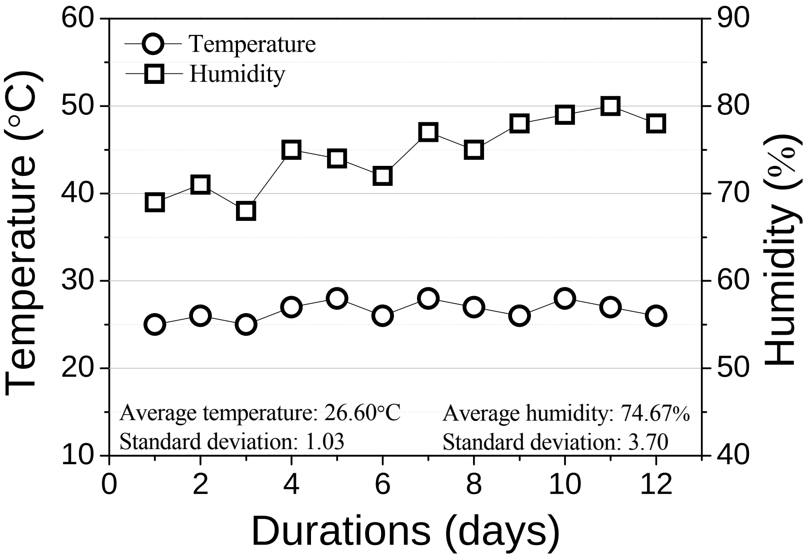


Fig. 4 Room temperature and humidity

Both locally available clay and quartz sand were dried at 105°C for 24 hours in oven (101-A, Leao, China). The clay was then mixed with different amounts of quartz sand (clay/sand = 2 : 8, 4 : 6, and 8 : 2) in a bottle for the regulation of average particle size of mixture, and then shaken until a uniform appearance was observed. After that, the soils were compacted in improved rigid-wall compaction-mold permeameter (TST-55, China) using the standard JGS 0162. Subsequently, all specimens were submerged into the DIW inside a vacuum chamber (BHG, Jingkeyusheng, China) connected to a pump (vp-0.5, Yangyi, China) for saturation. To ensure sufficient saturation, the saturation process lasted 36 hours under a vacuum with the pressure of -85 kPa.

Hydraulic conductivity tests were performed by the constant-head technique following the JIS A 1218 standard procedure. First, the specimens were permeated with DIW to get initial stable hydraulic conductivity. Second, the DIW was replaced by the bacterial culture Escherichia coli, which was incubated in a LB medium in 37°C for 24 h, and then poured into the soil with a certain water head following Shaw et al. (1984). Note that the specimens were always in a state of saturation and there were no air bubbles in this process. During the injection process, liquids were sampled at both the inlet and the outlet for measuring bacterial concentration. The identical bacterial concentration between the inlet and the outlet demonstrated the bacteria was evenly distributed in the specimens and the inoculation process was completed. Finally, the CM was used as the permeant liquid to flush the inoculated specimens. During the flushing stage, the filtrate was collected at the outflow vent every 12 hours and the outflow volume, duration, and hydraulic gradient were recorded for use in the hydraulic conductivity calculation following Darcy's law shown below:

$$k = \frac{L \cdot Q}{H \cdot A \cdot \Delta t} \tag{1}$$

where Q (mL) is the rate of the flow in time Δt (s), A is the cross-sectional area of the soil sample

(cm^2), L is the height of the soil sample (cm), and H is the hydraulic head difference (cm). Meanwhile the biomass concentration of the filtrate was measured via a colony counting method. The flushing stage terminated when the k of the specimens tended to be stable. To prevent variations of the conditioning parameters, the hydraulic conductivity tests were carried out in a temperature- and humidity-controlled room. As shown in Fig. 4, the indoor temperature and the humidity were maintained at 26.6°C and 74.7%, respectively, according to the measurement by Temperature/humidity meter (HTC-1, China). After the hydraulic conductivity tests, 1 g soil from upper, middle and lower parts of soil samples were collected for measuring the soil microbial biomass using the plate count method.

2.4 Numerical simulation model

The contaminant transport coupled model was established in the COMSOL software, and the groundwater flow module in the physical field was selected. Modern landfills commonly require a leachate collection system (LCS) over a low permeability composite liner to control the escape of contaminants from the landfill (Yu and Rowe, 2012). The drainage layer, a part of LCS, is represented usually by granular materials with high permeability. Under this circumstance, bioclogging becomes difficult to create. Thus, a schematic diagram of contaminant advection-dispersion through the biofilm-modified soil liners beneath the landfill sites was shown in Fig. 5, where h is the leachate head acting on the liner system and L is the thickness of the landfill liner system. The concentration of contaminant in the leachate was assumed constant. The mathematical model was developed mainly on the basis of the following assumptions: (1) landfill liner systems were both saturated and homogeneous; (2) contaminant migration was one-dimensional along the direction perpendicular to the plane layers of landfill liner systems; (3) adsorption was a linear and equilibrium process.

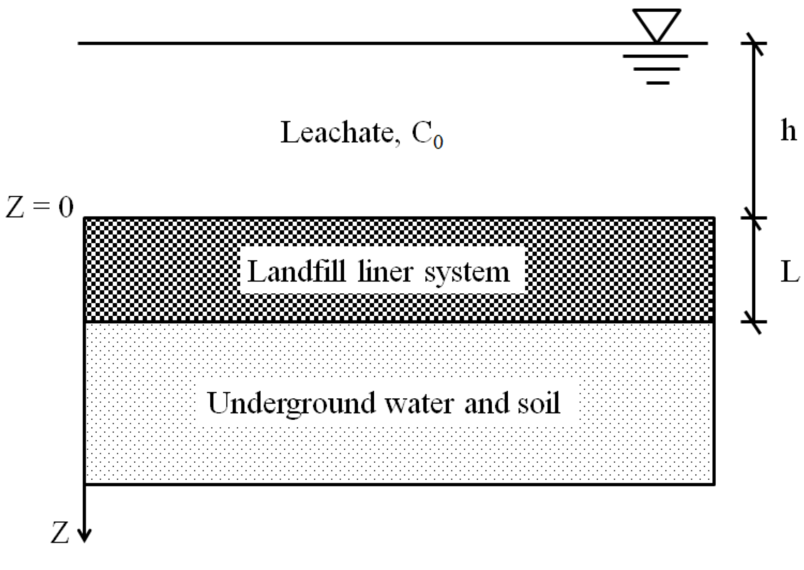


Fig. 5 Schematic diagram of mathematical model of contaminant transport through liner systems

The governing equation for contaminant transport through landfill liner systems was (Rowe et al., 2004):

$$R_d = \frac{\partial C}{\partial t} = D\frac{\partial^2 C}{\partial z^2} - v\frac{\partial C}{\partial z} \tag{2}$$

where t is time (a), C is the concentration (mass per unit volume of fluid) (mg/L), R_d is the retardation factor, which assumes a linear equilibrium sorption, z is the direction in which the diffusion occurs (m), D is the diffusion-dispersion coefficient of the solute in the mobile fraction (m^2/s), which includes both hydrodynamic dispersion and molecular diffusion (Rabideau and Khandelwal, 1998), v is the average linear seepage velocity (m/s).

Eq. 2 was subject to a variety of boundary conditions. Following Rabideau and Khandelwal (1998), this study assumed an initially uncontaminated barrier and adopted a conservative approach of prescribed contaminant concentrations at the edges of the barrier:

$$C(z,0) = 0 \quad (0 \le z \le L, t = 0) \tag{3}$$

$$C(0,t) = C_0 \quad (z = 0, t > 0) \tag{4}$$

$$\mathrm{C}(\infty, \mathrm{t}) = 0 \quad (\mathrm{x} = \mathrm{infinity}, \mathrm{t} > 0) \tag{5}$$

where C_0 is the concentration within the contained area (mg/L), L is the barrier thickness (m), t is time (a) and x is the distance from the upstream solution–sample interface (m). Although the conditions within the contained area may change with time, a reasonable and conservative initial approximation was to assume a constant source concentration (Eq. 4). According to Foose (2002), this assumption facilitated the development of the analytical solutions for contaminant transport through composite liners. Additionally, a zero concentration gradient was assumed at the bottom boundary (Eq. 5). Obviously, the migration distance of pollutants was limited, and it cannot diffuse indefinitely. This study focused on the effect of different factors upon the breakthrough time, which was defined as the downstream concentration of an indicator pollutant reaching 10% of the initial concentration of the leachate (Peter and Smith, 2002). Note that the breakthrough time should be greater than 30 years for a landfill site (Chen et al. 2015).

3. Experimental results and discussions

3.1 Hydraulic conductivity

The hydraulic conductivities of specimens permeated by DIW were shown in Table 2. During the DIW permeation process, the hydraulic conductivity of the three mixtures (specimen #1-3) reduced to 74%, 78% and 71%. This time-dependent decrease of hydraulic conductivity was attributed to the dilution effect of the leached out salts. Along with the permeation process, the leached out soluble salts became less, which caused the increase of the thickness of the diffusion double layer (DDL) and a decrease of the concentration of the outflow in response (Tang et al., 2014). Thereby, the lower concentration of outflow resulted in lower hydraulic conductivity (Shackelford et al., 2000). As shown in Table 2, the hydraulic conductivity k

decreased significantly with the increase of the clay content when permeated with DIW. When the sand content of the specimens increased, its gradation was harmful. A small amount of clay particles were not enough to fill the sand pores and resulted in the large hydraulic conductivity.

Table 2 Hydraulic conductivity of specimens permeated by DIW (Unit: cm/s)

Specimen	Initial *k*	Final *k*	Minimum *k*	Maximum *k*	Stable *k*
20%	7.2×10^{-5}	5.2×10^{-5}	5.1×10^{-5}	7.8×10^{-5}	5.3×10^{-5}
40%	5.5×10^{-6}	7.5×10^{-6}	7.4×10^{-6}	9.7×10^{-6}	7.6×10^{-6}
80%	7.9×10^{-6}	5.4×10^{-6}	5.3×10^{-6}	1.0×10^{-5}	5.6×10^{-6}

When live bacterial cultures passed through these specimens, the hydraulic conductivity of these specimens decreased sharply to 2.8×10^{-5} cm/s, 6.8×10^{-6} cm/s and 4.8×10^{-6} cm/s within a few hours. After live bacterial cultures were replaced by CM, all of specimens exhibited time dependent behaviors as shown in Fig. 6a. In general, *k* decreased almost one order of magnitude after 12 days of hydraulic conductivity test, and the specimen #3 achieved the lowest and most stable values of *k*. As indicated in Fig. 6a, the final stable *k* for specimen #1 (3.1×10^{-6} cm/s) was nearly 10 times less than the original value. For specimens #2 and #3, the *k* of clay-sand mixture decreased by 81% and 95%. The decrease of *k* due to bioactivity was attributed to the presence of biofilms (Francisca et al., 2010). Shaw et al. (1985) reached the same result, which was shown in Fig. 7a (I). The direct SEM examination of the inlet faces of the cores plugged by the live organisms showed that a thick and confluent amorphous film completely filled the pore spaces, and further reduced the permeability of this solid matrix sharply. Biofilms are communities of bacteria attaching and developing on surfaces embedded in a matrix of extracellular polymeric substances (EPS) (Carrel et al., 2018), the structure of which was shown in Fig. 7a(II). Biomass formation is a primary product of the microbial growth and the EPS excretion (Bottero et al., 3013). The biomass utilizes available nutrients from the surrounding environment for cell growth and secretion of EPS (Rowe, 2005). Thus, the presence of CM containing nutrients was responsible for the formation and stimulation of biofilms by Escherichia coli (Carter et al., 2016). When the hydraulic conductivity test continued, the soils would have more biofilms, and thus a greater biomass of bacteria would contain within those biofilms (Phillips et al, 2011).

As seen from Fig. 6a, at early stage of the experiment (1st - 2nd days), the *k* was not significantly reduced. There was a trend of the *k* reduction at the middle stage (3rd - 6th days). During the late stage (7th - 12th days), the decreasing rate of *k* retarded and tended to be stable. The S-shaped curve of biofilm development proposed by Bryers and Charakalis (1982) well explained this phenomenon. As shown in Fig. 7a (III), the biofilm development was the net result of the following transport and biological reaction.

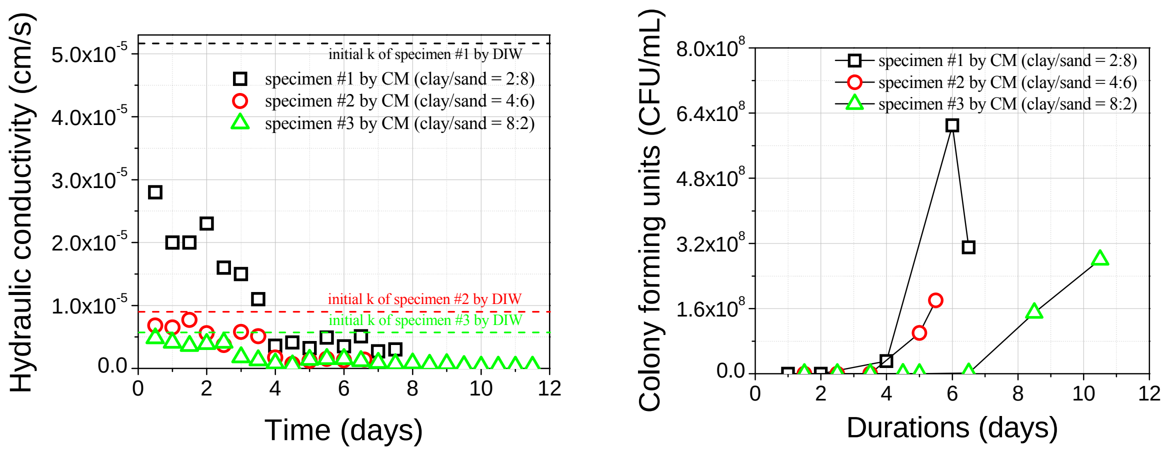
1) Lag phase: the bacterial cell adhesion to a solid was a two-stage process: reversible adhesion followed by irreversible adhesion. Biofilm accumulation rates in this phase

may be limited.

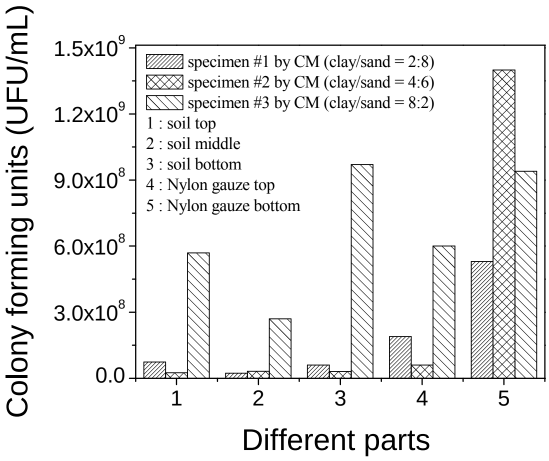
2) Exponential growth phase: the biomass was rapidly accumulated during this stage. Biofilm production was the net accumulation of attached material due to cellular reproduction and microbial production of extracellular polymers.

3) Plateau phase: during biofilm development, portions of biofilm peeled away from the surface were entrained in the fluid flow, which kept the total biofilms relatively constant.

According to Ryder et al. (2007), EPS provided a remarkably large surface area per unit volume for microorganism binding, which had a significant influence on attaching the soil surface and then led to floc formation by agglomeration of bacteria that partially blocked the soil pores (Sponza, 2002). The EPS also acted as a shield to prevent biofilm damage (Ryder et al., 2007). Obviously, biofilms are the best mode of bacterial life under these harsh conditions which are characterized as flowing, extremely low pH and high content of metal ions (Li and Sand, 2017). Inside the soil, the inoculated bacterium formed several layers of mesh forms of biofilm between soil particles as well as on the soil surface. The formation mesh layers effectively clogged the pore of soil and resulted in hydraulic conductivity reduction (Kim, 2004). As shown in Fig. 7b, this explanation was consistent with the observed permeability that declined in inoculated specimens after a certain period of permeation. Moreover, it is worth mentioning that polysaccharides can induce the formation of surface cracking, which are known to promote water infiltration. Finally, EPS can trap airborne particles, resulting in the accretion of new sand and clay layers that may increase the sorptivity of the soil, but may decrease the overall soil porosity (Rossi et al., 2012).

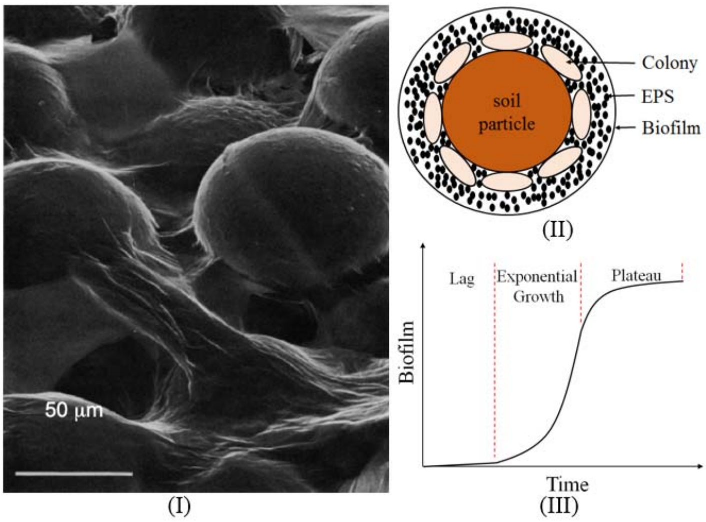


(a) (b)

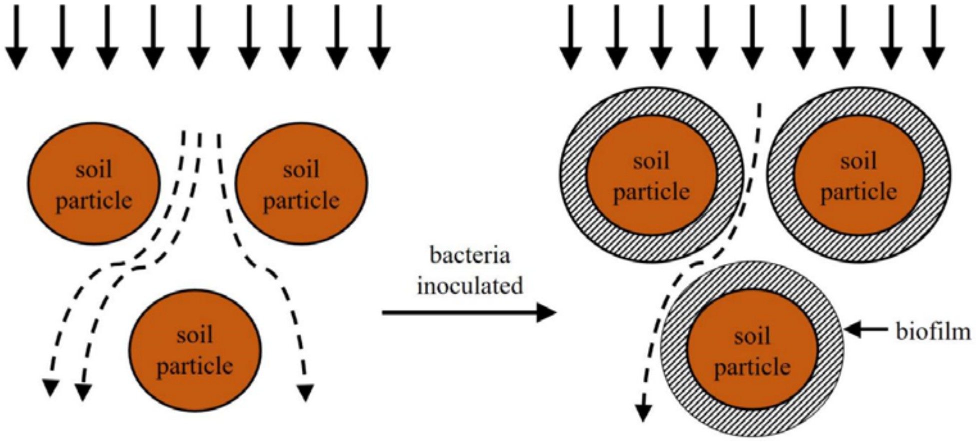


(c)

Fig. 6 Results of microbial permeation experiments. (a) The effect of microorganism on hydraulic conductivity of soil specimens; (b) The colony forming units of filtrates; (c) The colony forming units of soil.



(a)



(b)

Fig. 7 Biofilm mechanism. (a) Biofilm: (I) Scanning electron micrographs showing top surface of glass bead core encased in biofilm; (II) The structure of biofilm; (III) Idealized model of biofilm growth and accumulation; (b) Influence of biofilms on hydraulic conductivity.

3.2 The biological activity in filtrate

The bacterial concentration of filtrate, collected from the outlet ports in different times, was plotted in Fig. 6b. Prior to the injection test, the concentration of feed liquids was recorded. Bacteria detected in feed liquid of specimen #1 was 2.7×10^5 CFU/mL and specimen #2 and #3 were both 1.4×10^8 CFU/mL. During the initial stage of permeation tests, the colony forming units of filtrate was far less than feed liquid. This indicated two possible reasons: 1) in the early permeation experiment, the microorganism had been inoculated into the soil and its metabolic system needed to adapt the new environment; 2) some microorganisms were successfully injected into the soil specimens. Microbes attached to the surface of soil particles and developed biofilms. Previous studies on biofilms have shown that signaling chemical molecules released by the bacteria in biofilm affected the growth mode of bacteria inducing attached growth rather than suspended growth (Ren et al., 2013; Cha et al., 2008). Attachment was a complex progress affected by the growth of medium, substratum, and cell surface. The solid-liquid interface between a surface and an aqueous medium (e.g., water, blood) provided an ideal environment for the attachment and growth of microorganisms (Donlan, 2002). According to Seki and Mayzaki (2001), in the first stage of colony formation, bacteria attached to the solid particle, first reversibly and then irreversibly with EPS. Fimbriae are classical surface organelles of Gram-negative bacteria. Since their discovery by electron microscopy in the 1940s, fimbriae have been identified on Escherichia coli, Salmonella, Shigella, Proteus, Pseudomonas and other members of the Enterobacteri-aceae family (Duguid and Old, 1980). Mucosal pathogens used fimbriae for adherence to increase their colonization efficiency (Bergsten et al., 2007). However, some studies indicated that the non-pathogenic bacteria Escherichia coli had a low adherence and the biofilm coverage was low (Zhu et al., 2017). The low adherence of the bacteria was associated with its lack of a functional fim operon encoding type 1 fimbriae (Connell et al., 1996).

To overcome this limitation, Escherichia coli has been genetically transformed to express type 1 fimbriae to increase the capacity for adherence (Bergsten et al., 2007).

As shown in Fig. 6b, the microbial biomass rapidly increased in the latter stage. This suggested that the microbe had been adapted to the new environment and entered the logarithmic growth phase in which the microbial growth rate reached the maximum. It was demonstrated from Figs. 4a and 4b that the hydraulic conductivities of these specimens decreased rapidly and were the lowest during the logarithmic growth phase. Such phenomenon was caused by the formation of the biological clogging in the pore of clay-sand mixtures.

The adhesion capacity of microorganism towards surface was enhanced significantly with the clay content, which was validated in Fig. 6b. The initial filtrate colony forming units of specimen #1 was 34% of feed liquid, the initial filtrate concentration of specimen #2 and #3 was three and five orders of magnitude smaller than that of the inlet feed liquid. According to Table 1, the specific surface area of the clay used in this study was 24.49 m^2/g, which was higher than 2.5 m^2/g of a typical soil from the Chaco-Pampean plain used by Francisca et al. (2010). The clay content of specimen #3 was the highest, which resulted in the largest retention of microorganism in the soil. According to Donlan (2002), the soil surface had an important role in the attachment progress, and the extent of microbial colonization appeared to increase as the surface roughness increases. This was because shear forces diminished and surface area was higher on rougher surfaces.

3.3 The biological activity in different parts of specimens

The number of microorganisms in different parts of soil was obtained, which was shown in Fig. 6c. Among the three specimens, microbial biomass increased with time. The high biomass zone was mainly distributed at the bottom of the soil samples. While the low biomass zone was mainly located on top of the soil samples. Compared with other parts, the biomass at the bottom of soil and nylon gauze had substantial advantages. In particular, the microbial biomass at the bottom of nylon gauze were 1-2 orders of magnitude higher than the other parts. Such high biomass at the bottom was because that the bottom soil was firstly contacted fresh medium with a large and adequate contact surface, and the microbe grew rapidly at that location. With the development of permeation process, the soil pores were blocked by microorganisms. The biological clogging was notable and resulted in narrower pore spaces, which might restrict bacteria movement and activity, limit nutrient transport, diminish space availability, slow the rate of division, and finally lead to reduced bioactivity on top of the soil (Fredrickson et al., 1997). It was apparent that with the increase of clay/sand ratio, more microbial biomass formed in soil. If the microbial size was much smaller than the pore size of the soil, the CM would flush the soil with a high flow rate and the microbe could be carried away from the soil with a high water permeability.

3.4 Finite element analysis

In this study, a set of 14 different cases of contaminant diffusion results were presented in Table 3. As illustrated, Cases 1, 2 and 3 presented the effect of hydraulic conductivity in liner systems on the contaminant breakthrough time. Fig. 8 compared the effect of hydraulic conductivity of barriers on contaminant breakthrough time. To facilitate the comparison, a horizontal line representing a concentration of 0.1 C_0 was drawn as a baseline. It was seen from Table 3 that the decrease of the hydraulic conductivity from 1×10^{-8} m/s to 1×10^{-10} m/s increased the breakthrough time from 29.2 years to 130 years. Meanwhile, as shown in Fig. 8, after the same operation time, the pollutant concentration at the bottom of the barrier decreased with the decrease of the hydraulic conductivity. These results showed that low permeability soils were critical to reduce or retard the movement of leachate over time, and thereby protected water resources.

Cases 4, 5 and 6 were used to assess the effect of landfill liner thickness on the barrier performance. As presented, the performance of the liner system was significantly affected by the thickness. Table 3 indicated that the increase of the liner thickness from 0.6 m to 2 m increased the breakthrough time by 8.5 times. This may be because that more tortuous pathways experienced by solutes transporting through the thicker liners.

Cases 7, 8 and 9 were designed for investigating the effect of leachate head. As shown in Table 3, the breakthrough time was 95.8 years for contaminant when the leachate head above liners was 0.3 m (Case 7). The breakthrough time decreased to 56.4 years when the leachate head was 1 m (Case 8). When the water head reached 10 m, the breakdown time was shortened to only 10.8 years (Case 9). The results showed that the leachate head had a great impact on the performance of the liner system. Therefore, it was of great importance to restrict the leachate head to a relatively low value (e.g. <1 m), and further to reduce the diffusion of pollutants into the surrounding environment.

Cases 10 - 11 showed the effect of diffusion on breakthrough time of landfill liner system. It can be seen that the increase of the diffusion coefficient from 2×10^{-10} m^2/s to 2×10^{-9} m^2/s resulted in the variation of solute breakthrough time from 95.8 years to 13.7 years, which decreased by 85.7%. This indicated that breakthrough time decreased with the increase of diffusion coefficient.

Cases 12 to 13 investigated the influence of retardation factor on the breakthrough time. When the retardation factor increased from 1.1 to 1.5, the breakthrough time increased from 29.2 years to 39.6 years. The retardation factor had a great influence on the breakdown time. A higher value of R indicates that the soil is more capable of retarding the breakthrough of the contamination (Knop et al., 2008).

Table 3 Comparison of the breakthrough time of contaminant

Cases	K (m/s)	h (cm)	D (m^2/s)	R_d	L (m)	t (year)
Hydraulic conductivity						
1	1×10^{-8}					29.2
2	1×10^{-9}	30	2×10^{-10}	1.1	2	95.8
3	1×10^{-10}					130
Liner thickness						
4					0.6	10.1
5	1×10^{-9}	30	2×10^{-10}	1.1	1.2	37.2
6					3	95.8
Leachate head						
7		30				95.8
8	1×10^{-9}	100	2×10^{-10}	1.1	2	56.4
9		1000				10.8
Diffusion coefficient						
10	1×10^{-9}	30	2×10^{-10}	1.1	2	95.8
11			2×10^{-9}			13.7
Retardation coefficient						
12	1×10^{-8}	30	2×10^{-10}	1.1	2	29.2
13				1.5		39.6

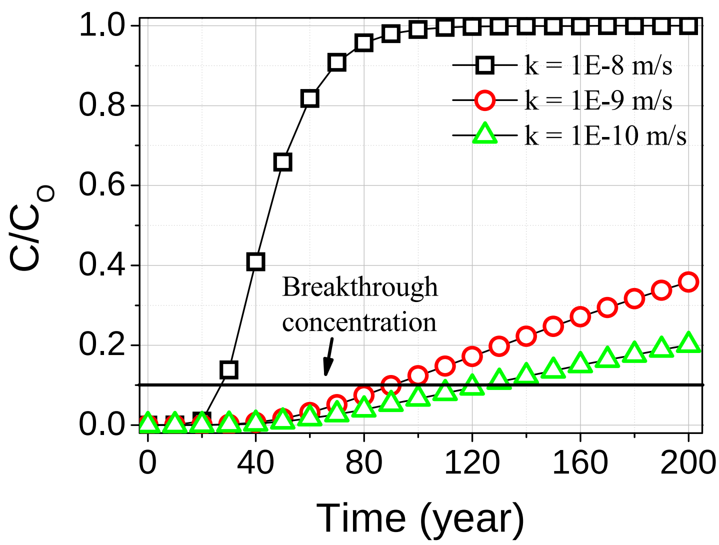


Fig. 8 Effect of hydraulic conductivity in barriers on contaminant breakthrough time

Development of a bio-mediated soil improvement technique requires an application strategy. The two primary strategies are built on bioremediation techniques, such as bioaugmentation (where the required microbes are injected into the soil) and biostimulation (where natural microbes are stimulated), developed over the last 30 years. Among them, the former has been the primary strategy used to date in exploring geotechnical applications (Dejong et al. 2013). To implement biological clogging into landfill engineering practice for reducing the permeability of liner material, the high pressure jet grouting technology can be applied to inject microorganisms into the soil, which makes soil and microbes fully contact. The treatment was applied at depths of original compacted clay liner at the bottom of the landfill site. The treatment involved injection of sufficient bacterial suspension that was cultivated in the laboratory. Biofilm formation, and the production of other extracellular polymeric substances (EPS), are additional biogeochemical processes that can impact on soil behavior. These processes generate organic solids that occupy a portion of the pore space with a soft, ductile, elastomeric-like material, and thereby reduce hydraulic conductivity.

The disadvantage to biofilm sealing technologies is the maintenance of established barriers; microbial biofilms will degrade over time without nutrient supply (Philips et al., 2013). However, a novel strategy using leachate leakage-driven biofilminduced "Intelligent self-repair" would be considered. When leachate leaks through an underlying clay liner or the defects in a geomembrane formed composite liner, biologically clogging of porous medium occurs. This is not only due to the early injected microbes' growth stimulated by the nutrients in the leachate, but also because of the natural microbes contained in the leachate. In addition, the deposition of organic and inorganic suspended particles also increased the clog mass accumulating in the soils (Yu and Rowe, 2012b). The majority of the studies on microbial geotechnology at present are at the laboratory level. Due to the complexity, the applications of microbial geotechnology would require an integration of microbiology, ecology, geochemistry, and geotechnical engineering knowledge (Ivanov and Chu, 2008). If degradation over time is of concern, the retreatment, or ‘healing’, of the biogeochemical treatment might be a possible solution (Montoya, 2012).

4. Conclusions

This study evaluated the effect the microorganism on hydraulic behaviors. The hydraulic conductivity tests were conducted by utilizing a modified permeameter. The specimens were fabricated by clay and sand with clay/sand ratios of 2:8, 4:6, 8:2, respectively. The microorganism Escherichia coli. was selected for pretreatment towards compacted soil specimens. Subsequently, the one-dimensional advection-dispersion model was developed for contaminant transport through liners. To evaluate of the barrier performance, a set of numerical simulation cases were investigated. The main conclusions were as follows:

(1) The presence of biomass in the soil specimens significantly decreased the hydraulic conductivity. The effective porosity, which contributed to fluid flow through soil liners, was significantly reduced due to the biofilm in the soil pores.

(2) The surface roughness inside the internal pores was enhanced with the increase of the clay content. This improves the adhesion between microorganism colonization and particle surface.

(3) Decreasing the hydraulic conductivity from 1×10^{-8} m/s to 1×10^{-10} m/s resulted in the increase of breakthrough time by 345.2%. Meanwhile, the significant retardation of contaminant was unable to observe in the fast flowing systems. Therefore, a low hydraulic conductivity was essential for the liner systems to achieve desirable barrier performance.

(4) Bioclogging of soils could be used in landfill engineering to reduce the permeability of liner material. However, to adopt the microbial method effectively, an integration of engineering, microbiological, and ecological studies and design consideration are required.

Acknowledgements

The study is supported by the National Nature Science Foundation of China (51708377), Natural Science Foundation of Jiangsu Province (BK20170339), Natural Science Fund for Colleges and Universities in Jiangsu Province (17KJB560008), project from Jiangsu Provincial Department of Housing and Urban-Rural Development (2016ZD18, 2017ZD002), and Jiangsu Provincial Transport Bureau (2016T05).

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