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Effect of sulfate erosion on strength and leaching characteristic of stabilized heavy metal contaminated red clay

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Abstract: Solidification/stabilization (S/S) technology has been widely used for remediation of the heavy metal contaminated soils. The heavy metal ions will be leached from the stabilized contaminated soil under sulfate erosion conditions, which gives rise to secondary contamination to the areas around the mine sites. The commonly used Portland cement, fly ash and quicklime were taken as binder raw materials with various mix proportions. And then, the sulphuric acid and nitric acid method was used to investigate the leaching characteristic of stabilized heavy metal contaminated soils. The effects of binder types and binder contents, sulfate concentrations (1.5, 3.0 and 6.0 g/L) and erosion time (0, 7, 14 and 28 d) on leached concentrations of heavy metal ions from contaminated soils were studied. Moreover, a parameter named immobilization percentage (IP) was introduced to evaluate the influence of erosion time and sulfate concentration on immobilization effectiveness for heavy metal ions. The results showed that, the leached heavy metal concentrations increased with sulfate concentration and erosion time. Comparatively speaking, the composite binders that had calcium oxide in it exhibited the worst solidification effectiveness and the lowest immobilization percentage, with the largest leached heavy metal concentration.

Key words: solidification/stabilization; heavy metal contaminated soil; sulfate erosion; sulphuric acid and nitric acid method

1 Introduction

Exploitation of non-ferrous metal mines unavoidably causes environmental problem of heavy metal contamination to the soil and groundwater around the mine area. The heavy metal contaminations in such cases are mainly from mineral processing, effluent discharge, tailings and solid waste storage [1,2]. Previous study shows that lead (Pb) and zinc (Zn) contaminations in the mining area occupy a leading position [3,4], which is one of the most serious threats to the fitness of humans, animals and plants. For example, Pb affects the skeletal and intellectual development, and Zn leads to refractory anemia [5].

As for remediation of the heavy metal contaminated soils, solidification/stabilization (S/S) technology is widely used to solidify the heavy metal ions from migrating to surrounding areas and to strengthen the contaminated soils [6,7]. By contrast to other remediation technologies, S/S technology has the advantage of low cost, easy construction and excellent blocking capability to biodegradation [8].

The raw materials of S/S for heavy metal contaminated soil are usually high alkaline cementhardening materials such as Portland cement, quicklime and fly ash [9], whose incorporation adjusts and changes the physicochemical properties of contaminated soil by physical absorption, chemical precipitation, coordination and oxidation-reduction. The chemical form of heavy metal ions in the contaminated soil can be changed from unstable state to stable state, and the migration and bioavailability of heavy metals are lowered to attain the remediation objective.

Sulphate attack is one of the common erosion phenomena, which could be caused by acid rain infiltration and industrial waste discharge etc and could lead to higher concentration of sulfate ion in groundwater. The acidic groundwater promotes the leaching of heavy metals like lead (Pb), zinc (Zn), cadmium (Cd) and manganese (Mn) from solidified/stabilized contaminated soil suffering sulphate attack [10]. Based on element

Foundation item: Project (41472278) supported by the National Natural Science Foundation of China; Project (2015B071) supported by the Beijing Nova Program, China; Projects (53200859533, 53200859536) supported by the Fundamental Research Funds for the Central Universities of China samples, many researchers studied the leaching and hydraulic characteristics of solidified/stabilized heavy metal contaminated soil under the influence of acid rain, by using semi-dynamic leaching test [10–17], infiltration test and soaking test [18–21].

KAMON et al [18] deeply investigated the erosion mechanism of lime and cement stabilized soil subjected to acid rain through infiltration with soaking test and concluded that the engineering and physico-chemical properties of the stabilized soils would change during a long time of acid rain erosion. STANFORTH and QIU [22] studied the solubility of lead contaminated soil treated by phosphate-based additive, coming to the result that the addition of soluble phosphate greatly reduced lead solubility. MALVIYA and CHAUDHARY [11] investigated the immobilization effectiveness of various heavy metals (Pb, Zn, Cu, Fe and Mn) using diffusion leaching tests and geochemical modelling for cement solidified/stabilized hazardous sludge, concluding that the leaching process of heavy metals from solidified/stabilized soils was controlled by pH condition and metal hydroxide solubility. MOON et al [12-14] conducted a series of modified semi-dynamic leaching tests for quicklime/fly ash treated contaminated soil and evaluated the solidification/stabilization effectiveness with diffusion coefficient and leachability indices of the target heavy metals. DALMACIJA et al [23] investigated the long-term leaching behavior of Pb- and Cdcontaminated sediment by performing modified semi-dynamic leaching tests using acetic acid and humic acid solution as leachant, and the standard toxicity characteristic leaching procedure (TCLP) showed that S/S samples turned to be acceptable for "controlled utilization". SONG et al [15] performed semi-dynamic leaching test, toxicity characteristic leaching procedure (TCLP) and sequential extraction procedures, in which concentrated sulfuric acid (H₂SO₄) or sodium hydroxide (NaOH) solution was taken as leachant, and investigated the leaching characteristics of heavy metals in cementbased solidified/stabilized sewage sludge. More recently, DU et al [10,19] and JIANG et al [20] studied the effect of acid rain with various pH on leaching and hydraulic characteristics of cement-based solidified/ stabilized lead contaminated clay through infiltration and soaking tests, obtaining the results that the leached concentration and leaching rate of calcium were significantly influenced by pH of acid rain and the sulfate ions in it. The above-mentioned studies indicate that the controlling mechanism of leaching of most heavy metals from S/S treated contaminated soil is diffusion [12,14,23], while in some cases, the mechanism controlling heavy metals leached from stabilized soils is surface wash-off or wash-off and diffusion combination [11,13,24].

There have been abundant research achievements

about leaching behaviors of heavy metal contaminated soils treated by solidification/stabilization, however, studies on mechanical and leaching characteristics of stabilized contaminated soils suffering secondary acid attack are still rare [18, 20–22].

The objective of this work is to investigate the influence of sulfate erosion on mechanical properties and leaching characteristics of cement based and composite cement/quicklime/fly ash based solidified/stabilized lead/zinc contaminated soil. A series of toxicity characteristic leaching procedures (TCLP) with sulphuric acid and nitric acid method were performed on Pb- and Zn-contaminated red clay that had been eroded to different extents. The influence of various possible factors including cement/quicklime/fly ash contents and proportions, concentration of sulfate solution, and erosion time of the solidified/stabilized soil in sulfate solution, on the leaching characteristics and mechanical properties of the soil was investigated, which can provide a basis for understanding the resistance capacity of binders for sulfate erosion.

2 Materials and method

2.1 Materials

The uncontaminated red clay used in this test was collected from the surrounding area of a nonferrous metal mine in Chenzhou City, China. The basic physical and chemical properties of the soils are summarized in Table 1. The water content was tested using gravimetric method as per HJ 613–2011 [25]. The Atterberg limits were tested as per GB/T 50123–1999 [26]. The pH of the soil was measured as per NY/T 1377–2007 [27]. The optimum water content and maximum dry density of the soil were determined as per the standard Proctor compaction test [26]. The chemical compositions of the uncontaminated red clay used in this work were determined by X-ray diffraction experiment, as listed in Table 2.

Property	Value
Natural water content, $w_n/\%$	27.3
Specific gravity, $G_{\rm s}/({\rm g}\cdot{\rm cm}^{-3})$	2.75
Plastic limit, w _P /%	22.6
Liquid limit, $w_{\rm L}/\%$	43.5
pH	7.93
Optimum water content, w _{op} /%	24.4
Maximum dry density, $\rho_{d,max}/(g \cdot cm^{-3})$	1.79

The commonly used cement, fly ash and lime were chosen as the stabilizers. Cement used in the tests was ordinary #42.5 Portland cement produced in Tianjin, China, with a specific surface area of $370 \text{ m}^2/\text{kg}$. Fly ash

used in the tests was obtained from Tianjin Guohua Co., Ltd., China, with a water requirement of 98%, ignition loss of 5.78% and surface area of 516 m²/kg, as secondary ash. Lime (CaO) used in the tests was obtained from Tianjin Fuchen Chemical Co., Ltd., China. The chemical constituents of the cement and fly ash were determined by X-ray diffraction experiment, as listed in Tables 3, 4 and 5. Besides, chemicals obtained from Beijing Chemical Works (China) including lead nitrate (Pb(NO₃)₂), zinc nitrate (Zn(NO₃)₂) and sodium sulfate (Na₂(SO₄)) were used in the tests.

 Table 2 Chemical compositions of soil used in this work (mass fraction, %)

SiO ₂	Al_2O_3	Fe_2O_3	MgO	CaO	Na ₂ O	TiO ₂	Others
65.22	13.35	7.5	1.567	1.425	1.223	0.340	9.36

 Table 3 Chemical compositions of cement used in this work (mass fraction, %)

CaO	SiO ₂	Al_2O_3	SO_3	Fe ₂ O ₃	MgO
44.7	27.4	13.1	3.9	3.34	1.19
MnO	K ₂ O	TiO ₂	Na ₂ O	P_2O_5	SrO
0.07	1.1	0.5	0.34	0.1	0.1

 Table 4 Chemical compositions of fly ash used in this work

 (mass fraction, %)

SiO ₂	Al_2O_3	Fe ₂ O ₃	CaO	MgO	SO_3	Loss	f-CaO
57.62	28.59	4.21	3.46	0.64	0.40	5.78	0.35

 Table 5 Particle size distribution of cement and fly ash used in this work

Particle size/µm	Fly ash/%	Cement/%
~2.40	21.74	11.51
~4.97	19.09	9.95
~10.28	15.95	17.63
~21.28	17.24	30.85
~34.56	11.68	19.37
~44.04	5.12	5.95
~56.13	4.18	3.19
>56.13	5.00	1.55

2.2 Specimen preparation

As Zn and Pb were commonly encountered in the contaminated sites worldwide, especially in China [10], they were selected as the target heavy metals in this work. To prepare mixed Zn- and Pb-contaminated soils, the air-dried clean red clay soil was soaked in a predetermined volume of lead nitrate and zinc nitrate solution for 7 d until the soil was thoroughly exposed to the heavy metal ion with chemical reaction. Then, the contaminated soils were air dried and crushed to pass through a sieve with opening size of 0.075 mm. The concentrations of lead ion and zinc ion of the artificially

contaminated soils are both 2000 mg/kg. Nitrate solutions were chosen as the chemical source of contamination because they have high solubility and are inert to cement hydration [28].

The binders for solidification/stabilization (S/S) used in the tests were prepared with different combinations and proportions of Portland cement (PC), fly ash and quicklime. A total of six types of binders were prepared, and the binders were designated as CiFjQk to denote specimens with individual Portland cement concentration of i%, fly ash concentration of j% and quicklime concentration of k%, as given in Table 6. The contents were measured on dry soil mass basis.

 Table 6 Components and proportions of binders used in this work

Binder	Components and proportions
C10	10% cement
C10F5	10% cement, 5% fly ash
C10F5Q2	10% cement, 5% fly ash, 2% quicklime
C15	10% cement
C10F10	10% cement, 10% fly ash
C10F5Q5	10% cement, 5% fly ash, 5% quicklime

Sulfate is commonly found in atmospheric, soil and groundwater environment, which may influence the solidification effect of contaminated soils exposed to it. Previous experimental investigations have indicated that if contaminated soils stabilized by high-alkali cementitious materials, such as Portland cement (PC), quicklime and pulverized fly ash, suffered acid rain or sulfate attack for a long time, heavy metals in the stabilized soils would be leached easily [10]. In this work, four types of sodium sulfates were prepared with the contents of 0, 1.5%, 3% and 6%, and the corresponding standing periods are 0, 7, 14 and 28 d, respectively.

To investigate the effect of acid environment on leaching behavior of stabilized Zn- and Pb-contaminated red clay, four sets of samples (three parallel samples in each set) were prepared with the following procedures:

1) The Zn- and Pb-contaminated red clay, cement, fly ash and quicklime were dried and crushed, the soil was ground to pass through a sieve with opening size of 2 mm for the soil, whereas cement, fly ash and quicklime were ground to pass through 0.5 mm sieve.

2) The optimum water content and maximum dry density of the soil were determined as per the standard Proctor compaction test [26].

3) The dried contaminated soil, distilled water and binders were fully mixed using an electronic mixer in accordance with the optimum water content and designed proportions listed in Table 7. The mixtures were compacted into cylindrical steel tube molds with 50 mm in diameter and 50 mm in height. Prior to the soil filling in the mold, vaseline was uniformly applied to the inner wall of molds. In each specimen, the heavy metal contents were kept the same, and the filling was performed in five stages in which each filling was followed by a 2 min shaking to ensure that the entrapped air bubbles were removed. Then, both ends of the specimen were covered with a steel lid, and the specimen was carefully extruded from the mold using a hydraulic jack.

4) The stabilized soil specimens were kept in the standard curing room (curing temperature (20 ± 3) °C, relative humidity 95%) for 28 d. Then, the specimens were soaked in sodium sulfates with different contents of 0%, 1.5%, 3% and 6%, and remained without disturbance for 0, 7, 14 and 28 d, respectively.

2.3 Testing methods

The total volume and mass of the specimens with different sulfate concentrations and various erosion time were measured and the total density was calculated. Then, the unconfined compressive strength of the samples was tested. After that, a small amount of the crushed soils was used to measure its water content, and the dry density of the sample was calculated according to the measured values of the total density and the water content. Another part of the crushed soils was air dried and ground to pass through 2 mm sieve and then used for determining pH of the sulfate eroded stabilized soil as per NY/T 1377–2007.

The remaining broken samples were ground and put through a sieve with opening size of 9.5 mm. The leachability of heavy metals from the stabilized soils was determined using the sulphuric acid and nitric acid method [29]. In this method, the extraction solution was made up of concentrated sulfuric acid and nitric acid mixture (mass ratio of sulfuric acid to nitric acid was 2:1), with pH value of 3.2±0.05 and solid-to-liquid ratio of 1:20. After the leaching toxicity test, a collection of the leachate was gathered and let sit for 1 h before testing the pH by using a HORIBA pH/COND METER D-54 pH tester. After the pH test, the leachate was filtered through a 0.45 µm membrane and its supernatant (about 10 mL) was taken. The pH value of the collected supernatant was acidized to less than 2 by concentrated nitric acid. Finally, the flame atomic absorption spectrometry was used to measure the concentrations of leached Zn and Pb [30], in which triplicates were made and the average values of concentration were recorded.

3 Test results

3.1 Dry density

Figure 1 shows dry density (ρ_d) of heavy metal contaminated soils stabilized by C10, C10F5 and

C10F5Q2 soaked in erosion solutions with different sulfate concentrations for 7 d. It can be found that when the sulfate concentration was relatively low (0–3.0 g/L), ρ_d of the stabilized soil increased with the sulfate concentration, whereas, when the sulfate concentration exceeded 3.0 g/L, ρ_d of the stabilized soil tended to decrease with increasing sulfate concentrations. Besides, C10F5Q2 stabilized soil had a higher ρ_d than that of the soils treated by other two binders. The difference of ρ_d between soils stabilized by the three types of composite binders became more pronounced with increasing sulfate concentrations.



Fig. 1 Variation of dry density with erosion concentration

Figure 2 illustrates the change of ρ_d of the stabilized contaminated soil eroded by sulfate of 6.0 g/L with erosion time. In the initial erosion stage, ρ_d increased with the erosion time and reached its maximum ($\rho_{d,max}$) at 14 d of erosion for the soils stabilized by C10 and C10F5, while for C10F5Q2 stabilized soil, $\rho_{d,max}$ was achieved at 7 d of erosion. After that, ρ_d went down with increasing erosion time.



Fig. 2 Variation of dry density with erosion time

3.2 Potential of hydrogen (pH)

Figure 3 presents variation of soil pH (at 28 d erosion time) with erosion concentration, indicating that

pH of the stabilized soil decreased with increasing erosion concentration. From Fig. 4, it can be seen that, for stabilized soils eroded by sulfate (6.0 g/L) for less than 14 d, the longer the erosion time, the lower the soil pH. After that, the soil pH grew with erosion time.



Fig. 3 Variation of soil pH with erosion concentration



Fig. 4 Variation of soil pH with erosion time

3.3 Unconfined compressive strength (q_u)

Figure 5 shows the variations of unconfined compressive strength (q_u) of stabilized soils with curing/erosion time, under standard curing condition (with sulfate erosion) and sulfate erosion condition (3.0 g/L), respectively.

It is evident that q_u of stabilized contaminated soil increased with curing time in standard curing condition. q_u of contaminated soil stabilized by C10F5Q2 was always higher than that of soils stabilized by C10 and C10F5 in every curing stage (Fig. 5(a)). In the initial curing stage (3–14 d), q_u of soil stabilized by C10 and C10F5 increased slowly by 2.2% and 1.5%, respectively, while for the C10F5Q2 stabilized soil, q_u increased by 15% in the same stage. At the late curing stage (14–28 d), q_u of the stabilized soils increased faster, and the final q_u of C10F5Q2 stabilized soil was 1.5 times higher than that of soils stabilized by C10 and C10F5. When the stabilized soils were eroded in sulfate environment (3.0 g/L), q_u of the treated soils increased first and then decreased with growing erosion time (Fig. 5(b)). Generally, the same variation was observed for the heavy metal contaminated soils treated by C10, C10F5 and C10F5Q2. In the initial erosion stage (3–14 d), q_u of the soils stabilized by these three composite binders increased with erosion time; in particular, q_u increased fast in the first 3 days. Then, q_u tended to decrease when erosion time exceeded 14 d.



Fig. 5 Variation of q_u with curing time: (a) Without secondary erosion; (b) Eroded in sulfate concentration of 3.0 g/L

3.3 Leached Pb and Zn

Figure 6 shows the variations of the leached Pb concentration with the erosion time for different sulfate concentrations, obtained from the leaching toxicity tests. It can be seen from Fig. 6 that Pb concentrations increased gradually with erosion time. In the early days of erosion, hydration reactions of cement, fly ash and quicklime led to adsorption and inclusion of Pb in the soil. However, as the erosion went on and corrosion effects increased, a portion of the absorbed Pb in the stabilized soil diffused into the solution. In regard to the leached amount of heavy metal ions for three types of composite binders, C10F5Q2 stabilized soil exhibited the largest amount of leached Pb while C10 stabilized soil showed the lowest.



Fig. 6 Variation of Pb concentrations leached from stabilized contaminated soil with erosion time: (a) 1.5 g/L sulfate solution; (b) 3.0 g/L sulfate solution; (c) 6.0 g/L sulfate solution

Figure 7 presents the variations of the leached Pb concentration with sulfate concentrations for different erosion time. As the sulfate concentration increased, the leached Pb concentration grew on the whole.

Figure 8 illustrates the variations of the leached Zn concentration with erosion time for different binder contents. It was observed that the higher the binder contents, the lower the leached Zn concentrations. The hydration products of the binders absorb and enclose the

heavy metal ions, more binders mixed will absorb and enclose more heavy metal ions, thereupon, less heavy metal ions can be leached.



Fig. 7 Variation of Pb concentrations leached from stabilized contaminated soil with sulfate concentrations: (a) 7 d erosion time; (b) 28 d erosion time

3.4 Immobilization percentage of heavy metal ions

To evaluate the solidification/stabilization effectiveness, a parameter named immobilization percentage (IP) was introduced [31], which is defined as

$$I = \frac{C_0 - C_1}{C_0}$$
(1)

where *I* refers to the immobilization percentage of heavy metal ions at a given erosion concentration and erosion time; C_0 refers to the concentration of heavy metal ions leached from untreated contaminated soil; C_1 refers to the concentration of heavy metal ions leached from stabilized soil at a given erosion concentration and erosion time.

Figure 9 shows the variations in IP for heavy metal ions with erosion time for different erosion concentrations. It can be seen that IP first increased and then decreased with erosion time.



Fig. 8 Variation of Zn concentrations leached from stabilized contaminated soil with erosion time: (a) 1.5 g/L sulfate solution; (b) 3.0 g/L sulfate solution; (c) 6.0 g/L sulfate solution

Figure 10 shows variations in IP for heavy metal ions with erosion concentrations for different erosion time. It is indicated that for 7 d of erosion, the IP of heavy metal ions decreased with increasing erosion concentrations. For stabilized soils eroded for 14 d and 28 d in low erosion concentration, the IP of heavy metal ions decreased with increasing erosion concentration; when the erosion concentration exceeded 3.0 g/L, the IP approximately tended to be stable. With regard to these three types of composite binders C10, C10F5 and C10F5Q2, the IP corresponding to C10F5Q2 was the lowest, while that of C10 was the highest.



Fig. 9 Variation of immobilization percentage of heavy metal ions with erosion time: (a) 1.5 g/L sulfate solution; (b) 3.0 g/L sulfate solution; (c) 6.0 g/L sulfate solution

4 Discussion

The test results obtained in this work suggest that the leaching characteristics and mechanical properties of S/S treated heavy metal contaminated soils eroded by sulfate were considerably affected by sulfate



Fig. 10 Variation of immobilization percentage of heavy metal ions with sulfate concentrations: (a) 7 d erosion time; (b) 14 d erosion time; (c) 28 d erosion time

concentration, erosion time and type of solidification binders. This can be mainly attributed to the effect of sulfate on the stability of calcium silicate hydrate (CSH) and calcium hydroxide (Ca(OH)₂) [10,19]. With regard to the Pb contaminated soil solidified with cementitious binders such as cement, fly ash and quicklime, Pb can be encapsulated in cement matrices through the combination of adsorption, substitution and precipitation onto the Ca hydrates and/or CSH phase [19,31–35]. Sulfate erosion leads to reduction of pH and increase of Ca, Pb and Zn in stabilized heavy metal contaminated soil [19,36]. The leached Pb suppresses the cement hydration and pozzolanic reactions and hinders the formation of Ca(OH)₂/CSH [19,37]. Moreover, a part of Ca(OH)₂/CSH formed in the soil matrix gradually dissolves under acidic condition [12,34], and as a result, the soil buffering capacity and degree of pore spaces filled with CSH in the soil matrix go down [10]. In addition, leached Ca is one of the major factors controlling the strength of solidified contaminated/ uncontaminated soil [21], and the more the leached Ca is, the lower q_u the solidified soils have.

For the first three types of composite binders, the highest leached Pb concentration was found in leachate of C10F5Q2 stabilized soil whereas the lowest leached Pb concentration was found in leachate of C10 stabilized soil. This can be attributed to the presence of CaO in C10F5Q2. The hydration reaction of CaO created ettringite (3CaO·Al₂O₃·3CaSO₄·32H₂O), from which the Ca²⁺ replaced Pb²⁺ in sulfate erosion condition and Pb²⁺ leached into the solution. As a result, the C10F5Q2 stabilized heavy metal contaminated soils leached most heavy metal ions in the toxicity characteristic leaching procedures [38,39].

5 Conclusions

1) The dry density (ρ_d) of the contaminated soils solidified by C10, C10F5 and C10F5Q2 increased first and then decreased with increasing sulfate concentrations. The ρ_d of the C10F5Q2 solidified soil was slightly larger than that of C10 and C10F5 solidified soil, for the solidified soil eroded in 6.0 mg/L sulfate solution for 7 d. The ρ_d difference between the soils solidified by three types of composite binders was no more than 0.22%.

2) Under the sulfate erosion, pH of the solidified heavy metal contaminated soil decreased with increasing sulfate concentration. As erosion went on, the pH of solidified soil increased first and then decreased.

3) The unconfined compressive strength (q_u) of the solidified heavy metal contaminated soil gradually increased in the initial stage of sulfate erosion (3-14 d), while in the late erosion stage, q_u tended to decrease.

4) In the leaching toxicity test by sulphuric acid and nitric acid method, leached Pb concentrations increased with erosion time and sulfate concentrations.

5) Under the same erosion conditions, the amount of leached heavy metal ions from C10F5Q2 solidified soils was the largest, whereas that from C10 solidified soils was the smallest.

6) The leached heavy metal concentrations decreased with increasing binder contents. The highest immobilization percentage (IP) of heavy metal ions was

found in C10 solidified soils, the IP corresponding to C10F5Q2 solidified soils was the lowest and that corresponding to C10F5 solidified soils was the intermediate, indicating that the composite binder C10F5Q2 had the worst immobilization effectiveness to heavy metal ions under the erosion of sulfate.

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硫酸盐侵蚀条件下固化重金属污染土强度及浸出特性

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摘 要: 污染土修复常采用固化稳定化技术。固化稳定化重金属污染土在酸侵蚀条件下其重金属离子会重新溶出, 从而导致对周边环境的二次污染。以常用的水泥、粉煤灰和石灰为固化剂原料,设计不同组合及配比的固化剂, 通过无侧限抗压强度试验及硫酸/硝酸法毒性浸出试验研究固化重金属污染土在酸侵蚀条件下的强度及浸出特性。 探讨固化剂类型、硫酸盐浓度(1.5,3.0,6.0 g/L)和侵蚀龄期(0,7,14,28 d)对固化重金属污染土的强度及溶出浓度的 影响,并引入固定率参数,进一步量化考察侵蚀龄期和侵蚀浓度对重金属离子固定率的影响。结果表明:重金属 离子的浸出量随着硫酸盐侵蚀浓度和侵蚀龄期的增加而增加;相对而言,含氧化钙的固化剂对重金属污染土的固 化效果较差,重金属离子浸出量大,固化率低。

关键词:固化/稳定化;重金属污染土;硫酸盐侵蚀;硫酸/硝酸法

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