



## Application of accelerated carbonation with a combination of $\text{Na}_2\text{CO}_3$ and $\text{CO}_2$ in cement-based solidification/stabilization of heavy metal-bearing sediment

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### ABSTRACT

The efficient remediation of heavy metal-bearing sediment has been one of top priorities of ecosystem protection. Cement-based solidification/stabilization (s/s) is an option for reducing the mobility of heavy metals in the sediment and the subsequent hazard for human beings and animals. This work uses sodium carbonate as an internal carbon source of accelerated carbonation and gaseous  $\text{CO}_2$  as an external carbon source to overcome deleterious effects of heavy metals on strength development and improve the effectiveness of s/s of heavy metal-bearing sediment. In addition to the compressive strength and porosity measurements, leaching tests followed the Chinese solid waste extraction procedure for leaching toxicity – sulfuric acid and nitric acid method (HJ/T299-2007), German leaching procedure (DIN38414-S4) and US toxicity characteristic leaching procedures (TCLP) have been conducted. The experimental results indicated that the solidified sediment by accelerated carbonation was capable of reaching all performance criteria for the disposal at a Portland cement dosage of 10 wt.% and a solid/water ratio of 1:1. The concentrations of mercury and other heavy metals in the leachates were below 0.10 mg/L and 5 mg/L, respectively, complying with Chinese regulatory level (GB5085-2007). Compared to the hydration, accelerated carbonation improved the compressive strength of the solidified sediment by more than 100% and reduced leaching concentrations of heavy metals significantly. It is considered that accelerated carbonation technology with a combination of  $\text{Na}_2\text{CO}_3$  and  $\text{CO}_2$  may practically apply to cement-based s/s of heavy metal-bearing sediment.

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### 1. Introduction

Anthropogenic activities such as metal smelting, battery manufacture and municipal solid waste disposal give rise to very large quantities of heavy metal emissions each year. Heavy metals may cause contamination of water, soil or other receiving streams and then threaten the health of human beings, animals and plants [1]. If heavy metal contents in river sediment exceed environmental acceptable limits, cleanup action is warranted. Among remediation strategies for contaminated sediments, phyto-remediation, washing and cement-based solidification/stabilization (s/s) are most commonly used over the world at present [2,3].

The phyto-remediation utilizes plants to take up and enrich heavy metals by rhizosphere bioremediation. It takes longer reme-

diation time compared to other technologies, and may actually lead to leaching of toxins into the water during the planting process [4,5]. In addition, the products containing heavy metals need appropriate disposal to prevent the secondary pollution.

In the washing process for decontamination of sediment, acids or complexing ligands are used to extract heavy metals [6]. The selectivity of removing heavy metals can be improved and complications caused by the presence of other ions may be avoided in the subsequent operations by the use of tailor-made extracting agents. Dealing with effluent is, however, unavoidable. As metals are usually precipitated out of the effluent with an alkali to form sparingly soluble hydroxides, the sludge formed needs dewatering, and, upon passing leaching tests, long-term management, typically in an engineered landfill [5,6].

Cement-based s/s reduces the mobility of pollutants and subsequent hazards for the environment by a combination of solidification and stabilization. It has been used for the disposal of various types of industrial wastes, but is particularly suited for lands contaminated with heavy metal compounds for re-development [2,3]. It is also considered to be one of the most effective options for treating severely contaminated sediments in terms of time

Abbreviations: A, alumina,  $\text{Al}_2\text{O}_3$ ; C, CaO; F,  $\text{Fe}_2\text{O}_3$ ; H,  $\text{H}_2\text{O}$ ; S,  $\text{SiO}_2$ ; C,  $\text{CO}_3^{2-}$ ;  $\text{C}_3\text{S}$ , tricalcium silicate,  $\text{Ca}_3\text{SiO}_5$ ;  $\text{C}_3\text{A}$ , tricalcium aluminate,  $\text{Ca}_3\text{Al}_2\text{O}_6$ ;  $\text{C}_4\text{AF}$ , ferrite,  $\text{Ca}_2\text{AlFeO}_5$ ; C-S-H, calcium silicate hydrate gel.

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and cost at present. The common binders used in cement-based *s/s* are Portland cement and various blended cements. With an optimized binder formulation and suitable operation conditions, cement-based *s/s* cannot only achieve and retain the desired physical properties for improving the handling characteristics, but also chemically stabilize or bind contaminants to lower the leaching rates [7]. The control of the speciation of metals is of paramount importance in minimizing the leaching rates during the long-term management of the solidified waste. The mechanical strength and the leaching resistance are two basic parameters of evaluating the effectiveness of the *s/s* process and allow direct comparisons of disposal options to be made.

As heavy metals may have deleterious effects on the hydration of cement, accelerated carbonation by supplying CO<sub>2</sub> during the mixing process has been exploited. It is capable of inducing setting and strength development where hydration of cement would be significantly retarded [8–10]. Carbonation neutralizes the alkaline nature of cement-solidified sediments, reduces the solubility of certain heavy metals and represents a significant advantage in terms of environmental protection. McWhinney et al. noted that metals such as Ba, Cr, Pb, Cd and Hg promoted cement carbonation in the presence of CO<sub>2</sub> [11,12]. Lange et al. reported the beneficial effects of carbonating solidified metal wastes curing in CO<sub>2</sub>. For example, they reported improvements in 28-day compressive strength, lower leaching rates and reduction in porosity compared to non-carbonated analogous [9,10]. Walton et al. demonstrated that some radionuclides formed solid solutions with calcite during the carbonation process and led to significantly lower release rates in carbonated waste forms compared to non-carbonated analogous. They also found, however, that nickel, cadmium, mercury, lead and cobalt exhibited higher leaching rates from carbonated waste forms compared to non-carbonated analogues [13]. Hills et al. reported that Cr concentrations in the leachates of non-carbonated samples were generally higher than their carbonated analogues [8]. They also reported that Cu concentrations were similar for both carbonated and non-carbonated samples, and carbonated samples demonstrated higher Zn concentrations at low acid additions and comparable concentrations at higher acid additions compared to non-carbonated samples. These conflicting results could be caused by different operating conditions, leading to the differences in the degree of carbonation. Of course, it should be noted that the differences in chemical properties of heavy metals, waste characteristics, and leaching methods and conditions would all have influences on the effectiveness of the process.

When CO<sub>2</sub> is used as a carbon source, carbonation reactions are rapid acting on the surfaces of the cement grains and are often self-limiting. The carbonation products occlude the inner regions of each grain from the CO<sub>2</sub> source. Consequently, for less accessible inner phases, the reaction requires a relatively long time to occur, as carbonation of cement largely depends on the diffusion and hydration of CO<sub>2</sub>. The complete carbonation of cement with CO<sub>2</sub> is difficult to achieve even in laboratory with grinding [14]. This work applies sodium carbonate as an internal carbon source and CO<sub>2</sub> as an external carbon source for accelerated carbonation of the mixture of Portland cement and heavy metal-bearing sediment. The objectives of the research were to examine the effect of accelerated carbonation on heavy metal immobilization and to compare the efficacy of hydrated and carbonated *s/s* techniques.

**Table 2**  
The chemical and mineral compositions of raw materials (%).

Materials	SiO <sub>2</sub>	Al <sub>2</sub> O <sub>3</sub>	CaO	Fe <sub>2</sub> O <sub>3</sub>	SO <sub>3</sub>	MgO	K <sub>2</sub> O	Na <sub>2</sub> O	C <sub>3</sub> S	C <sub>2</sub> S	C <sub>3</sub> A	C <sub>4</sub> AF
PC	21.19	6.67	64.25	3.35	3.28	0.74	0.65	0.29	56	18	8	12
Sediment	28.47	9.26	23.59	5.71	1.96	1.75	0.94	0.65				

**Table 1**  
The content and leaching amount of heavy metals in the sediment.

Metals	Hg	Cu	Pb	Zn	Ni
Content, mg/kg	6.64	115.69	154.86	239.95	13.37
Leachate concentration, mg/L	0.41	7.06	10.38	18.30	0.68
Regulatory level <sup>a</sup> , mg/L	0.10	100	5	100	5

<sup>a</sup> Chinese Standard: GB5085-2007.

## 2. Experimental

### 2.1. Materials

Five sediment samples were collected from the landfill leachate treatment lagoon of a municipal solid waste facility. Following the USEPA digestion method (that is, acid digestion with concentrated H<sub>2</sub>SO<sub>4</sub> and HNO<sub>3</sub> in excess of K<sub>2</sub>MnO<sub>4</sub> at 96 °C for 2 h), the contents of heavy metals in the sediments were analyzed. The sediments contained (mg/kg) mercury (3–12), lead (84–166), copper (91–130), nickel (5–27) and zinc (140–450). The filtered sediments were mixed at the same dry weights and then the leaching characteristics of the mixed sediment were quantified based on the Chinese solid waste extraction procedure for leaching toxicity – sulfuric acid and nitric acid method (HJ/T299-2007). The extraction fluid was a dilute solution of sulfuric acid and nitric acid at a ratio of 2:1 (pH 3.20 ± 0.05). The leaching results are shown in Table 1. It can be seen that mercury and lead concentrations in the leachate also exceeded the limits of 0.10 mg/L and 5 mg/L, as set by the Chinese criteria for hazardous waste (GB5085-2007).

In this work, a commercial Portland cement (PC, 42.5 grade, Blaine specific surface 310 m<sup>2</sup>/kg) complying with Chinese standard (GB175-1999) was used as a binder. The chemical compositions of the cement and sediment are presented in Table 2. All chemical reagents used in the experiments were AR grade.

### 2.2. Methods

#### 2.2.1. Preparation of specimens

The filtered sediment, with moisture of 20%, was mixed with Portland cement and the additive (Na<sub>2</sub>CO<sub>3</sub> for accelerated carbonation) for the preparation of specimens. Portland cement dosages used for *s/s* of the sediment were 10%, 15% and 20% by dry weight. The treatment of the sediment was performed over a range of solid/water (*s/w*) ratios at 2:3, 1:1 and 3:2.

**2.2.1.1. Hydration.** Each fresh paste of Portland cement/sediment was cast into 3 prism molds (40 mm × 40 mm × 160 mm) on a vibrating table (frequency: 2800–3000 min<sup>-1</sup>, amplitude: 0.75 mm) to prepare specimens for the compressive strength measurements. All prisms were cured in sealed plastic bags for 3 days, 7 days, 14 days and 28 days at a temperature of 20 °C. To stop the hydration reactions of samples for leaching tests, following the method used by Poon et al. [15], the pieces resulting from strength tests were soaked in acetone for a total of 14 days but with the acetone changed after 7 days. Then they were dried at a temperature of 60 °C for 48 h in a vacuum oven.

**2.2.1.2. Carbonation.** Each fresh paste of Portland cement/sediment was mixed with 5% Na<sub>2</sub>CO<sub>3</sub> (by dry weight) and then cast into 3

prism molds at the same way as for the hydration experiments. The prisms with  $\text{Na}_2\text{CO}_3$  were de-molded for 1 day, and carefully placed into a carbonation chamber and exposed to a purity of  $\sim 100\%$   $\text{CO}_2$  at a pressure of 0.3 MPa for the same times as the hydration experiments. The carbonation reaction chamber contained a saturated NaCl solution to maintain relative humidity of 50% within it.

### 2.2.2. Compressive strength measurement

The compressive strength of three prisms of each mix was determined at 3, 7, 14 and 28 days. According to the requirement of Chinese standard, GB/T17671-1999, two ends of the specimens were ground to ensure that the surfaces were flat and parallel. The strength result reported was the average of three specimens with a variation of no more than 10%. If a specimen's variation of strength exceeded the average, the result would be the average of two specimens.

### 2.2.3. Mercury intrusion porosimetry (MIP)

The samples used for MIP tests in this work were the fractured, un-ground pieces of the above compressive strength test specimens (28 days of age). The mercury intrusion porosimetry was conducted with Auto Pore IV 9510 mercury intrusion porosimeter.

### 2.2.4. X-ray powder diffraction (XRD)

A Siemens D500 diffractometer and Kristalloflex 810 generator (Cu  $\text{K}\alpha$  radiation) were used to identify the crystalline products. The accelerating voltage was 40 kV and the current was 40 mA. The finely ground samples ( $<30\ \mu\text{m}$ ) were examined between  $5^\circ$  and  $40^\circ$   $2\theta$  at scanning rate of  $1^\circ$   $2\theta$  per minute. The diffractograms were obtained with Diffplus and analyzed using Bruker/AXS EVA software and were compared with the current version of the international powder data file (ICDD-JCPDS).

### 2.2.5. Leaching tests

The stabilized samples were subjected to three leaching test protocols: the Chinese solid waste extraction procedure for leaching toxicity – sulfuric acid and nitric acid leachant method (HJ/T299-2007), German leaching procedure (DIN38414-S4) and US toxicity characteristic leaching procedure (TCLP). In the case of HJ/T299-2007, samples of 10 g of crushed solidified sediment ( $<10\ \text{mm}$ ) was rotated on a roller at 30 rpm with 100 mL extraction liquid in a plastic bottle for 18 h at an ambient temperature of around  $20^\circ\text{C}$ . The preparation of extraction fluid involved mixing concentrated sulfuric acid and nitric acid (2:1) and then diluting to pH  $3.20 \pm 0.05$  with de-ionized water. For leaching tests with TCLP, extraction fluid 2 (acetic acid, pH 2.88) was used at a solid/liquid ratio of 1:20. The crushed solidified sediment ( $<10\ \text{mm}$ ) with the extracted fluid was rotated on a roller at 30 rpm for 18 h at an ambient temperature of around  $20^\circ\text{C}$  in a plastic bottle. For leaching tests with DIN38414-S4, 100 mL of distilled water was added to 10 g of crushed solidified sample ( $<10\ \text{mm}$ ) in a plastic bottle and was rotated on a roller at 30 rpm for 24 h at an ambient temperature of around  $20^\circ\text{C}$ . After leaching, the bottles were centrifuged and the supernatant liquids were filtered through a  $0.45\ \mu\text{m}$  membrane filter. The pH of the supernatant liquids was measured and the sample of the leachates was acidified for metal analyses using a dual view model PerkinElmer Inductively Coupled Plasma-Atomic Emission Spectroscopy. Mercury concentrations in the leachates were determined using cold vapor atomic absorption spectrometry. All leaching tests were performed in duplicate and all analyses of the leachates were conducted in triplicate. The results reported were the average values of measurements.

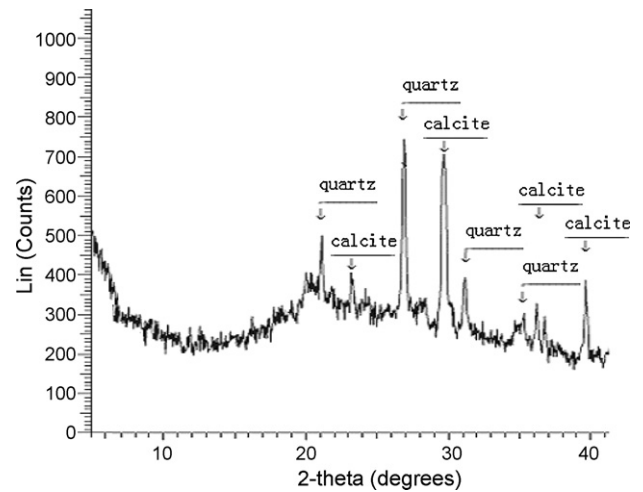


Fig. 1. The diffractogram of the sediment.

## 3. Results and discussion

### 3.1. Reaction product identification by XRD

Fig. 1 shows the diffractogram of the sediment, indicating the presence of quartz ( $21^\circ$ ,  $27^\circ$ ,  $31^\circ$  and  $35^\circ$ ,  $2\theta$ ) and calcite ( $23^\circ$ ,  $29.5^\circ$ ,  $36^\circ$  and  $39.5^\circ$ ,  $2\theta$ ). The weak peaks at other  $2\theta$  positions suggested the presence of minor kaolinite and montmorillonite.

Fig. 2 shows the diffractograms obtained from the solidified sediment at a cement dosage of 20% with s/w ratio of 1:1 at the age of 7 days and 28 days of hydration. The X-ray reflections of portlandite (CH) at  $18^\circ$ ,  $28.6^\circ$  and  $34^\circ$  ( $2\theta$ ), alite and belite at  $29.5^\circ$ ,  $32^\circ$ ,  $32.7^\circ$ ,  $34^\circ$  and  $38.5^\circ$  ( $2\theta$ ), and ettringite (AFt) at  $9^\circ$ ,  $16^\circ$ ,  $18^\circ$ ,  $22.8^\circ$ ,  $24.5^\circ$ ,  $25.5^\circ$ ,  $29.5^\circ$ , and  $32^\circ$  ( $2\theta$ ) can be identified. The peaks of quartz at  $26.5^\circ$  and  $35.5^\circ$  ( $2\theta$ ) and calcite at  $23^\circ$ ,  $29.5^\circ$ ,  $36^\circ$  and  $39.5^\circ$  ( $2\theta$ ) were also present. As would be expected, the peak intensities due to ettringite and portlandite increased with time.

It is worth noting that no independent crystalline precipitates of heavy metals were detected. In the literature, the possible mechanisms of metal immobilization in cementitious binders can be

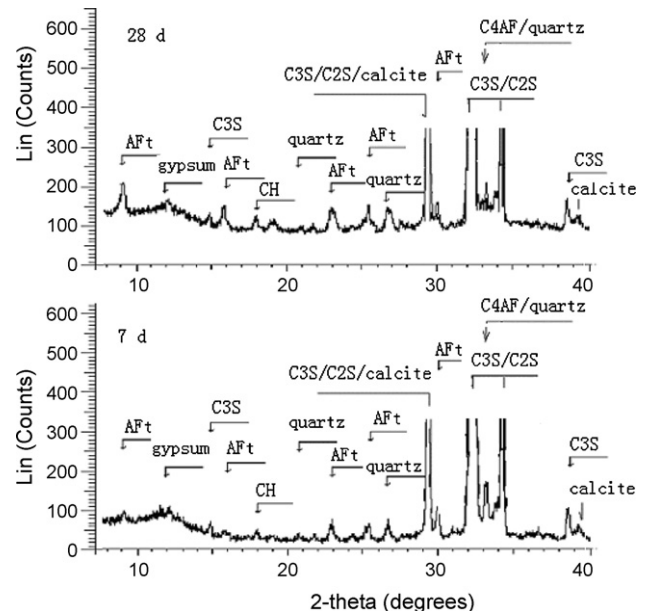


Fig. 2. Diffractograms of the hydrated solidified sediment.

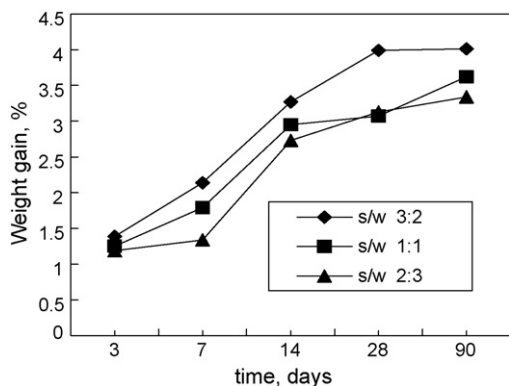
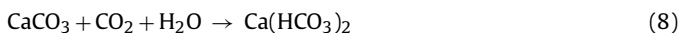
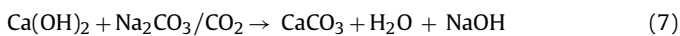
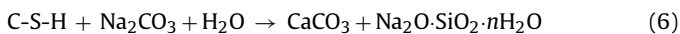
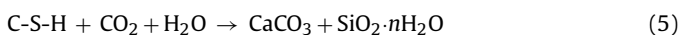
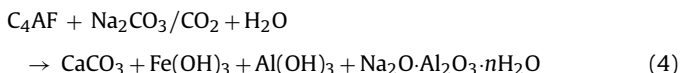
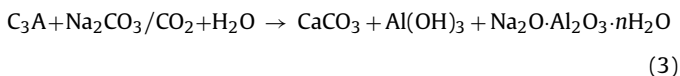
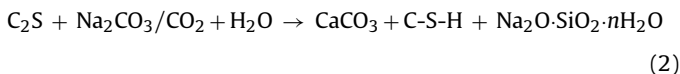
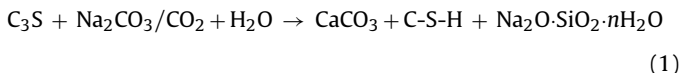


Fig. 3. The weight gain of the solidified sediment with time at different s/w ratios.

divided into physical adsorption, chemical adsorption, chemical incorporation, precipitation, and micro- or macro-encapsulation. In this study, heavy metals may be incorporated in hydration products of cement due to sorption or diadochy mechanism. It is unlikely to form independent precipitates of heavy metals, as heavy metal contents in the sediment were not high enough.

During the accelerated carbonation, cement phases react with  $\text{HCO}_3^-$  and  $\text{CO}_3^{2-}$  resulted from the dissociation of  $\text{Na}_2\text{CO}_3$  and hydration of  $\text{CO}_2$  to form carbonates and other products, for example, gelatinous hydrated silica, alumina, and iron hydroxide (see Eqs. (1)–(8)). In the carbonation process, C-S-H was subjected to decalcifying, initially by lowering of its Ca/Si ratio, and ultimately by conversion into  $\text{CaCO}_3$  and a highly porous, hydrous form of silica gel. In addition, portlandite reacted with  $\text{HCO}_3^-$  and  $\text{CO}_3^{2-}$  and changed into calcite.



The weight gain during the carbonation process reflects the degree of carbonation in cement and waste due to  $\text{CO}_2$  during curing. After a desired period of carbonation time, the samples were removed from the carbonation chamber and weighed. Fig. 3 presents the weight gain at a cement dosage of 20% with different s/w ratios. The weight gain was attributed to reactions of cement phases and hydration products with  $\text{CO}_2$ . It can be seen from Fig. 3, the weight of cement and sediment mixtures did not increase after a period of 28 days. This indicated that 28 days was suitable for curing of the solidified sediment.

Fig. 4 shows the diffractograms obtained from the solidified sediment at a cement dosage of 20% with s/w ratio of 1:1 at an age of

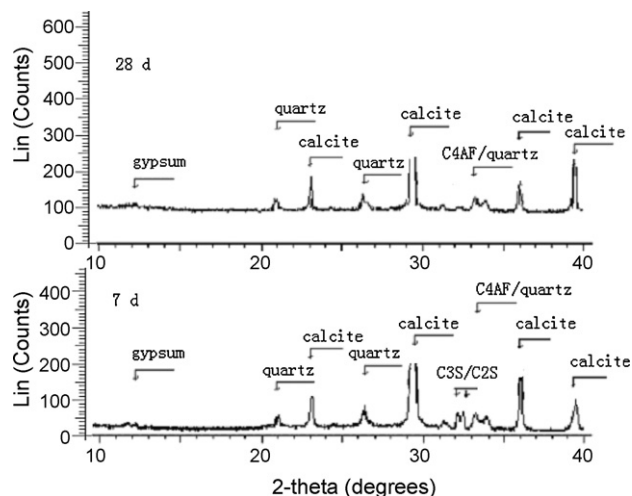


Fig. 4. Diffractograms of the carbonated solidified sediment.

7 days and 28 days of carbonation. The X-ray reflections for calcite at  $23^\circ$ ,  $25.5^\circ$ ,  $29.5^\circ$ ,  $36^\circ$  and  $39.5^\circ$  ( $2\theta$ ) were very strong and quartz was also detected. The typical double peaks around  $32.5^\circ$  ( $2\theta$ ) for alite and belite decreased significantly with time, as the degree of carbonation increased with time. At an age of 28 days, the X-ray reflections for alite and belite disappeared, indicating that the carbonation of alite and belite was complete. The reflections obtained for ferrite suggested this phase was resistant to carbonation. Analogous to hydration, no reflections related to metal containing compounds were detected in the carbonated solidified waste forms.

Yousuf and Mollah [16], and Valls and Vazquez [17] reported that heavy metals exhibited an affinity towards the cement reaction products mainly into nano-porous C-S-H gel, silica gel and fresh calcite by adsorption. These attractions primarily involve electrostatic force, Van der Waals force, hydrogen-bonding, and chemical bonding. It is certain that silicon-rich C-S-H gel resulted from cement carbonation favored the sorption of metal cations [6,14,18,19]. As pH and redox potential of systems influence these interaction forces, it is expected that the higher  $E_h$  values due to carbonation ( $-240$  mV to  $-270$  mV), compared to  $-300$  mV in hydration, would provide for the effective control of multivalent metals such as mercury at the higher oxidation states and favored the adsorption. Heavy metals such as lead, copper, zinc and mercury may be incorporated into calcium silicate hydrates or form solid solution with calcite, which accounts for its low solubility in the mild pH environment that formed in carbonation [20,21].

### 3.2. Compressive strength and pore structure

The compressive strength of the solidified sediment increased with curing time and the binder dosage, as shown in Fig. 5. Evidently, the compressive strength increased with the decreasing s/w ratios. For all ages, the compressive strength of carbonated samples was much higher than that of non-carbonated samples.

In hydration with s/w ratio of 1:1, the binder addition of 20% was capable of reaching  $1000 \text{ kN/m}^2$  (1 MPa) on the 28th day (US acceptable limit: 0.35 MPa, UK acceptable limit: 0.75 MPa). In carbonation, this can be achieved at lower s/w ratios or less additions of cement, indicating that accelerated carbonation can use cement more efficiently and can be operated at higher water contents. These are important advantages to the practical implementation of cement-based s/s for sediment treatments.

To investigate the mechanism by which the carbonation increased the compressive strength, the pore size and volume

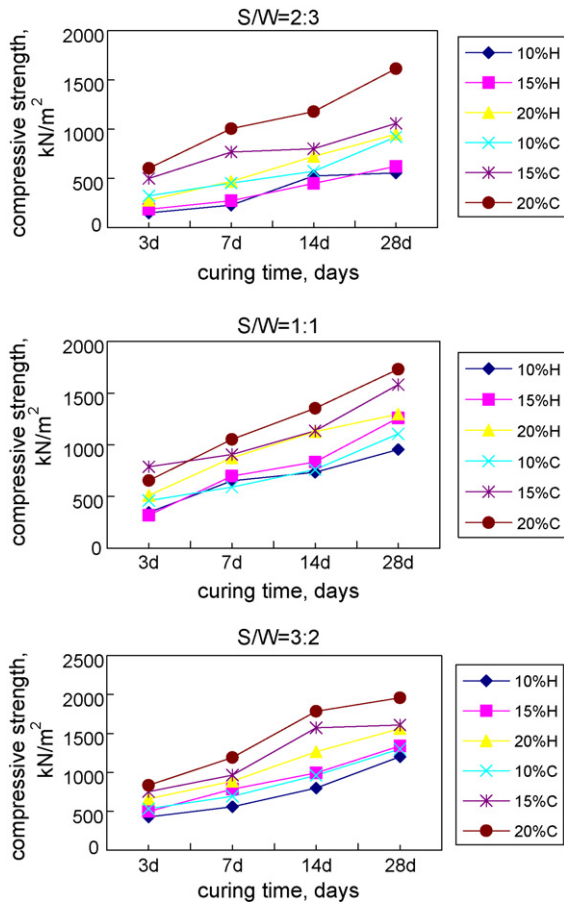


Fig. 5. Compressive strength development for solidified sediments at different binder dosages and s/w ratios (H stands for hydration, C for carbonation).

have been measured using mercury intrusion porosimetry. Fig. 6 presents the influence of carbonation on the pore size and volume of solidified sediments at a cement dosage of 20% with s/w ratio of 1:1. Compared with the hydrated sample, carbonation decreased the cumulative porosity greatly. This can be attributed to the precipitation of calcite in pores. The conversion of each gram mole of portlandite into  $\text{CaCO}_3$  results in an increase of 11.8% in volume and 35% in weight. The reduction of the porosity is the main reason that carbonation increased the compressive strength compared to hydration, as strength depends mainly on the porosity [22–26]. Additionally, carbonation can overcome the deleterious actions of insoluble metal colloidal gel and organic matters in the sediment, which may retard the normal hydration of cement and then the development of strength.

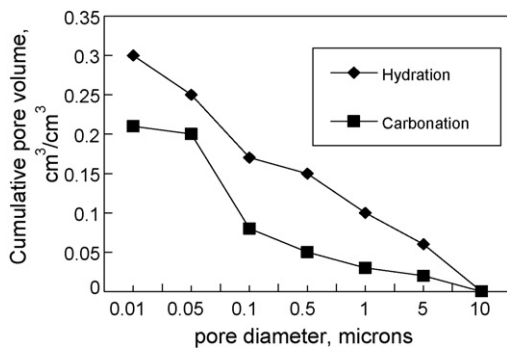


Fig. 6. Influence of carbonation on the pore size distribution at a cement dosage of 20% with s/w ratio of 1:1.

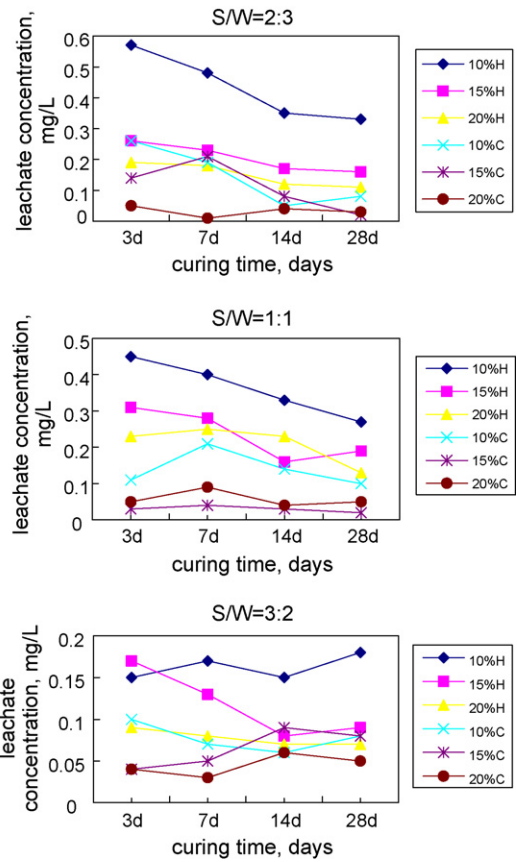


Fig. 7. Mercury leaching from treated sediments (H stands for hydration, C for carbonation).

### 3.3. Comparison of leaching test results

The variations of leachate concentrations of mercury with time shown by the solid waste extraction procedure for leaching (HJ/T299-2007) are presented in Fig. 7. It can be seen that s/w ratios exhibited a significant effect on metal leaching. The higher s/w ratio, the lower the mercury concentration in leachates. The accelerated carbonation can reduce mercury mobility greatly compared to the hydration. The leachate concentrations of carbonated samples were below the acceptable limit of 0.10 mg/L (Chinese standard: GB5085.3-2007), whereas in the hydrated samples, this limit can be hardly satisfied. This indicates that the solidified wastes by accelerated carbonation may have better resistance to acid rain attacks. The improvement of carbonation could be mainly attributed to the decrease of porosity, which decreased the permeability of the solidified sediment.

Table 3 presents mercury concentrations in the leachates as shown by the DIN and TCLP tests. It can be concluded that mercury concentrations in the DIN extracts of carbonated samples with 20% of Portland cement were below the acceptable limit (0.05 mg/L). As shown in Table 3, in the DIN tests, accelerated carbonation lowered the leachable amount of mercury compared to the hydration. This could be attributed to the decrease of porosity and the reduction of pH from 12.5 (hydration) to 10–11 (carbonation), which corresponds with minimum solubility of many compounds such as calcite and oxides in cement matrices and reduces the release of toxic metals [21]. In TCLP tests, however, the difference in the leachate concentration of mercury was very small between the hydrated and carbonated samples. The regulatory level (0.20 mg/L) can be satisfied only at higher s/w ratios. Despite the reduction of porosity and the addition of  $\text{Na}_2\text{CO}_3$ , accelerated carbonation

**Table 3**  
Mercury concentrations in leachates from the solidified sediment.

s/w ratio	Binder (%)	DIN tests, mg/L				TCLP tests, mg/L			
		7 days		28 days		7 days		28 days	
		Hyd <sup>a</sup>	Car <sup>b</sup>	Hyd <sup>a</sup>	Car <sup>b</sup>	Hyd <sup>a</sup>	Car <sup>b</sup>	Hyd <sup>a</sup>	Car <sup>b</sup>
2:3	10	0.29	0.15	0.32	0.07	0.55	0.53	0.44	0.28
	15	0.16	0.13	0.09	0.09	0.33	0.31	0.37	0.32
	20	0.09	0.04	0.11	0.04	0.11	0.12	0.34	0.27
1:1	10	0.14	0.12	0.12	0.06	0.21	0.16	0.17	0.14
	15	0.12	0.13	0.09	0.05	0.14	0.19	0.11	0.12
	20	0.09	0.08	0.11	0.04	0.16	0.19	0.12	0.13
3:2	10	0.12	0.09	0.19	0.07	0.31	0.21	0.23	0.13
	15	0.09	0.07	0.10	0.04	0.15	0.21	0.12	0.11
	20	0.04	0.07	0.05	0.04	0.11	0.10	0.16	0.12
RL <sup>c</sup> , mg/L		0.05				0.20			

<sup>a</sup> Hyd stands for hydration.

<sup>b</sup> Car for accelerated carbonation.

<sup>c</sup> RL for regulatory level.

**Table 4**  
Leaching of metals from the carbonated solidified sediment.

Element	Hg	Cu	Pb	Zn	Ni
DIN, mg/L	0.05	0.62	0.51	3.21	0.48
TCLP, mg/L	0.12	0.57	4.35	3.74	1.21
HJ/T299-2007, mg/L	0.07	0.48	4.27	5.59	2.06
Regulatory level <sup>a</sup> , mg/L	0.10	100	5	100	5

<sup>a</sup> Chinese Standard: GB5085-2007.

did not improve the resistance against organic acidic corrosion due to the decrease of Ca(OH)<sub>2</sub> contents. This may imply the dosage of sodium carbonate would need to be increased in the practical implementation to prevent solidified wastes from attack by organic acids.

Table 4 shows the leaching test results of heavy metals presented in the sediment following the Chinese solid waste extraction procedure for leaching toxicity (HJ/T299-2007), DIN and TCLP. Before leaching, the sediment was treated by accelerated carbonation with a combination of Na<sub>2</sub>CO<sub>3</sub> and CO<sub>2</sub> and cured for 28 days at s/w ratio of 1:1 and 15% addition of Portland cement. It can be seen that all metal concentrations were below the Chinese regulatory level (GB5085.3-2007), indicating that accelerated carbonation technology with a combination of Na<sub>2</sub>CO<sub>3</sub> and CO<sub>2</sub> may practically apply to cement-based s/s of heavy metal-bearing sediment.

#### 4. Conclusions

This work showed that cement-based s/s was capable of successfully remediating heavy metal-bearing sediment. The accelerated carbonation with a combination of dissolved Na<sub>2</sub>CO<sub>3</sub> and gaseous CO<sub>2</sub> can improve compressive strength of the solidified sediment significantly by reducing porosity and consequently permeability that would, in principle, be desirable for a matrix used to encapsulate toxic wastes. It was important to control factors such as s/w ratios, cement dosages and carbonation conditions, which affected strength development and leaching of heavy metals. These parameters should be optimized as they are also critical factors influencing the long-term performance of the solidified waste. The fixation mechanism of heavy metals requires further investigation and long-term monitoring is an absolute necessity.

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