



## ***In-situ* stabilisation followed by *ex-situ* composting for treatment and disposal of heavy metals polluted sediments**

YU Guangwei<sup>1,2</sup>, LEI Hengyi<sup>1,\*</sup>, BAI Tao<sup>1</sup>, LI Zhong<sup>1</sup>, YU Qiang<sup>1</sup>, SONG Xianqiang<sup>1</sup>

1. School of Environmental Science and Engineering, Sun Yat-sen University, Guangzhou 510275, China. E-mail: [ygwqlan@126.com](mailto:ygwqlan@126.com)

2. Department of Environmental Science and Engineering, South China Agriculture University, Guangzhou 510642, China

Received 12 August 2008; revised 21 October 2008; accepted 23 November 2008

### **Abstract**

An innovative *in-situ* stabilisation treatment followed by *ex-situ* sediment composting was tested for its ability to treat and dispose of heavy-metal-polluted sediments in a river near the Chinese Pearl Delta. First, polluted sediments were treated *in-situ* to stabilise the heavy metals. Then the treated sediments were dredged, dewatered and sent for high temperature aerobic composting (HTAC) treatment. Finally, the compost products were used as a fertiliser for river bank plants. The stabilisation efficiency of heavy metals during the process was investigated and the results are as follows: (1) using *in-situ* stabilisation, the extraction concentrations of Cu, Zn and Pb were reduced by 65.0%, 82.2% and 90.0%, respectively, which are much lower than the national standard given in the Identification Standard for Hazardous Waste (GB5085.3-1996); (2) chemical fraction analysis showed that heavy metals were further stabilized during the HTAC treatment; (3) the concentrations of Cu, Zn and Pb in rainwater leachate through the river bank met the level of class V in the Environmental Quality Standards for Surface Water in China (GB3838-2002). Therefore, using this new process, the toxicities of heavy metals in sediments were reduced markedly.

**Key words:** contaminated sediments; treatment and disposal; heavy metal; *in-situ* stabilisation; resource utilisation

**DOI:** 10.1016/S1001-0742(08)62357-8

### **Introduction**

In South China, the Pearl River Delta area is covered by a complicated urban river network where most of the rivers are tidal rivers and provide flood control and drainage. More importantly, these water bodies have received domestic sewage and industrial effluents for more than 50 years. As a result, over 90% of the rivers suffer from serious pollution, which mainly are organic contaminants, nutrients, and heavy metals. This pollution results in a sulphide odour problem, toxic sediment accumulation, and poor water quality. In Guangzhou, a typical delta city in the Pearl River Delta area, more than 230 urban rivers have serious problems with contaminated sediment deposition. The average thickness of the polluted sediments is about 50 cm, while the thickest is nearly 1.8 m (Li *et al.*, 2005). The deterioration of water quality over the entire river network has impacted the drinking water quality of this area.

In order to improve the water quality, the program called “The Comprehensive Improvement Project of the Pearl River Water Environment” was carried out by local government in 2002. The key goals are the river pollution treatment project control over black-odours and eliminating contaminated sediment. Dredging was employed to meet these two goals, while also increase the drainage

capacity of river. However, dredging was difficult to implement because of the cost of both dredging and treating the dredged sediments (hazardous wastes). Therefore, an alternative strategy, combining selected dredging and *in-situ* sediment treatment, was put forward.

This article reports a new process for the treatment and disposal of sediments polluted with heavy metals in an urban tidal river in Guangzhou City. First, polluted sediments were treated *in-situ* to oxidise sulphide and to stabilise heavy metals. Then, the treated sediment (non-hazardous wastes) was dredged, dewatered, and sent for high temperature aerobic composting (HTAC) treatment. Finally, the compost product was used as fertiliser for the river bank plants.

In recent years, *in-situ* treatment has gained attention because it is less expensive than *ex-situ* treatment or disposal of contaminated sediment (Jon, 1998). Generally, *in-situ* sediment treatment can be divided into two types as biological/chemical treatment methods and solidification/stabilisation treatment methods. The latter involves the addition of cements or chemicals to encapsulate contaminated sediments and/or convert them into less soluble, less mobile, or less toxic forms. For stabilisation of heavy metals, chemicals (e.g., lime and phosphates) are effective in both sediment and soil (Cotter *et al.*, 1996; Wang *et al.*, 2001; Hettiarachchi *et al.*, 2001; John, 2003). In

\* Corresponding author: E-mail: [leihengyi@126.com](mailto:leihengyi@126.com)

particular, heavy metal phosphate is effective in reducing the solubility or bioavailability of heavy metals, especially the phosphate-Pb composite, which was considered stable over a wide range of environmental conditions. Many *in-situ* sediment treatment projects employing biological/chemical techniques have been reported in the USA, Canada, Japan and other nations (Jon, 1998).

Composting is one of the most efficient and environmentally friendly methods for solid waste (e.g., sediments and sewage sludge) treatment and disposal (Aparna *et al.*, 2008). There are many advantages of using solid waste compost on land, including providing a whole array of nutrients for plant growth, reducing the need for fertilisers and pesticides, and improving soil physical and biological properties. However, this technique has been limited by enriched heavy metal content in the compost (Wei and Liu, 2005; Nomeda *et al.*, 2008). It is generally agreed that the total amount of heavy metals cannot fully reflect their ecological impact in soil environment, while their effects are more a result of the different chemical forms present than the total concentration (Hsu and Lo, 2001; Nomeda *et al.*, 2008; He *et al.*, 2008). Many studies on the speciation of heavy metals in composted sewage sludge have reported that, in general, heavy metals become less available during the composting process and composting tends to redistribute the metals from more labile forms to more fixed forms. However, some metals appear to behave differently depending on the source of the compost (Greenway and Song, 2002) and its initial chemical state (Petrizzelli *et al.*, 1994). Thus, it is worthwhile to analyze the chemical speciation of heavy metals during HTAC of sediments that have been conducted *in-situ*.

In this article, this new process was introduced and experimental results were reported, including *in-situ* stabilisation efficiency of heavy metals Cu, Zn and Pb, the dewatering capacity of the treated sediment, the chemical fractions of heavy metals in sediments before and after HTAC treatment, and the monitoring results of heavy metal concentrations in rainwater leachate of river bank soil with compost products.

## 1 Methodology

### 1.1 River and sediment samples

The Guocun River studied in this study, is located upriver of the Guangzhou City river network and has a mixed semi-diurnal tidal type (reflux period, a flux period, and an intertidal period). The treatment areas were 200 m long and averaged 25 m wide; the thickness of contaminated

sediment ranged from 0.5 to 1.5 m.

Prior to the *in-situ* treatment, the treatment areas were sampled for chemical and physical characteristics analysis, which are presented in Table 1. The sediments were heavily polluted by organic matter, nutrients, heavy metals, and sulphides. In addition, the extraction concentration of Pb was exceeded 3.0 mg/L, as given in the Identification Standard for Hazardous Waste-Identification for Extraction Toxicity (ISHW/GB5085.3-1996), indicating that these sediments should be treated as hazardous wastes. Therefore, before dredging, the sediments were pretreated to reduce the toxicity of the sulphides and heavy metals to improve the sediment dewatering capacity. For drainage purposes, the dredging sediment thickness was determined to be 30 cm.

### 1.2 Technology and process

The technology and process are shown in Fig. 1.

#### 1.2.1 *In-situ* treatment technology

The *in-situ* sediment treatment included oxidising sulphides by sediment aerating, stabilising heavy metals using phosphates, and improving the sediment dewatering capacity by injecting lime and flocculants. Raft Jet aerating boats (RJAB) were designed and used for *in-situ* sediment aeration and stabilisation.

There are three components of the RJAB system. (1) The aerating system, including the jet aerating system and the blower aerating system. The former is designed mainly for sediment mixing and aerating, while the latter increases dissolved oxygen in water columns without disturbing the sediments. (2) The chemical delivery system, which consists of chemical mixing and storage tanks, injection pumps, and associated controls to deliver the treatment reagents directly to the sediments after sediment aerating. (3) The operating platform, which is a raft used to store the equipment and chemicals and provide movement over the treatment area.

Sediments are exposed during low tide, which decreases the efficiency of mixing chemicals with sediments. Therefore, at the downriver of the treatment areas, an inflatable dam was used to maintain a water depth of 50 cm during low tide.

The *in-situ* treatment began with sediment aerating during low tide. Contaminated sediments were stirred and suspended in the overlying water by the jet aerating system of RJAB. In this case, 3 boats were performed 6 h per day and the total aerating capability of each boat was 20 kg O<sub>2</sub>/h. Stabilisation followed sediment aerating, where chemicals were added by the injecting system and

**Table 1** Characteristics of the sediment before treatment

	Organic matter (g/kg)	TN (g/kg)	TP (g/kg)	AVS (g/kg)	Redox (mV)	Heavy metals (mg/kg)		
						Cu	Zn	Pb
Mean	70.52	2.660	1.680	3.51	-230	2418	2074	150.4
Minimum	11.87	0.270	1.890	1.25	-260	428.6	358.7	175.9
Maximum	86.25	3.160	0.830	5.90	-130	5563	2436	94.45
Extraction concentration (mg/L) <sup>a</sup>						13.04 ± 0.61	29.54 ± 1.65	3.63 ± 0.23

TN: total nitrogen; TP: total phosphorus; AVS: acid volatile sulphide.

<sup>a</sup> By the extraction procedure for toxicity of solid waste-horizontal vibration method (GB5086.2-1997).

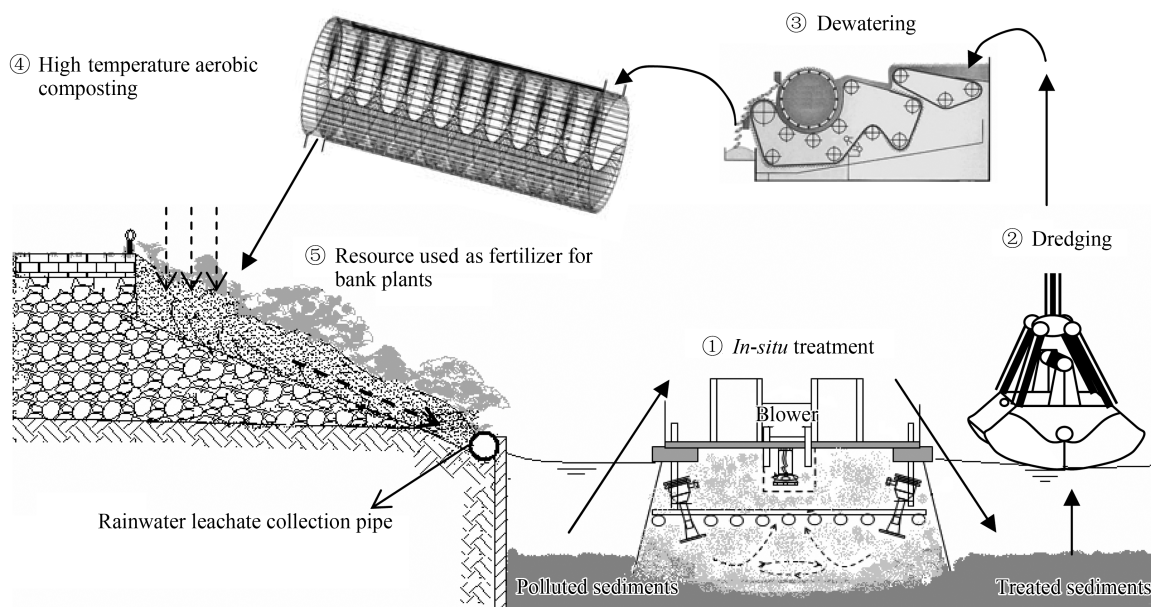


Fig. 1 Schematic of the process for *in-situ* stabilisation followed by *ex-situ* composting technology.

thoroughly mixed with the suspended sediment by the jet system. Every day, phosphates were added first and then the lime and flocculants at the end of treatment, which quickly settle the suspended sediment to avoid secondary-pollution resulting from suspended sediment during the flux period.

### 1.2.2 Ex-situ sediment treatment and disposal

The pretreated sediments (about 1500 m<sup>3</sup>) were dredged, mechanically dewatered, and sent to HTAC treatment directly after adding amendments (10%, volume), including wood chips, tangerine stems, expanding agents, beer brewing wastes, and thermophilic bacteria. The volume of each compost sample was 10 m<sup>3</sup> and the starting temperature was 30°C.

After HTAC treatment, the compost products were applied as fertiliser for river bank plants. The test area on the river bank was about 100 m<sup>2</sup> with a stone/cement floor, in which alternanthera philoxeroides and shrubs were planted to prevent soil erosion and improve the ecological system of the river bank. For this study, over 30 m<sup>3</sup> of compost product was added, as shown in Fig. 1. A PVC pipe ( $d = 10$  cm) was fixed at the bottom of the bank to collect the rainwater leachate and periodically measure the heavy metals concentration. The sampling frequency was 30 times per year.

### 1.3 Sampling and analysis

River sediments were collected using a glass tube with a height of 2 m and an inner diameter of 3 cm. To protect the sediment from oxidation, each sample was sealed with a plastic bag and immediately frozen. Dredged sediment and sediment materials for composting were collected with a grab sampler and were either processed on the day of the sample collection or immediately frozen and processed within two weeks.

The oxidation reduction potential (ORP) of the samples was measured. Analysis of sediment acid volatile sulphide (AVS) followed the method outlined by Brouwer and Murphy (1994). In this work, a 500-mL glass jar with a height of 14 cm and an inner diameter of 8 cm was used as a reaction vessel. First, a small 60 mL vial filled with 30 mL of 0.5 mol/L NaOH solution was glued to the inner wall of the jar. Then, the sample (5–10 g of wet sediment) was transferred to the vessel and 50 mL of 1 mol/L HCl was added. Next, within 4 h, the sediments were stirred regularly by a magnetic stirrer to ensure all sediments in suspension. Finally, the dissolved sulphide concentration in NaOH solution was measured via spectrophotometer at a wavelength of 670 nm.

The method of sequential extraction used in this study was developed by Tessier *et al.* (1979). Each chemical fraction was defined as follows. (1) Exchangeable: 1 g sediment (dry weight) was extracted with 8 mL of pH 7, unbuffered 1.0 mol/L MgCl<sub>2</sub> in Teflon centrifuge tubes for 1 h at 25°C with continuous agitation. (2) Carbonate: the residue from the exchangeable fraction was extracted with 8 mL of pH 5 1.0 mol/L NaOAc for 6 h at 25°C with continuous agitation; (3) Fe-Mn oxide: the residue from the carbonate fraction was extracted with 8 mL 0.04 mol/L NH<sub>2</sub>OH·HCl in 25% acetic acid (V/V) for 6 h at 96°C with continuous agitation. (4) Organic: the residue from the Fe-Mn oxide fraction was extracted with 2 mL of 0.02 mol/L HNO<sub>3</sub> and 3 mL of 30% H<sub>2</sub>O<sub>2</sub> at pH 2 for 2 h at 85°C with continuous agitation, and an additional 3 mL of 30% H<sub>2</sub>O<sub>2</sub> at pH 2 for 3 h at 85°C with continuous agitation; then, 3 mL of 3.2 mol/L NH<sub>4</sub>OAc in 20% HNO<sub>3</sub> (V/V) was added and agitated continuously for 0.5 h at 25°C. (5) Residual: the residue from the organic fraction was moved into beakers and digested with aqua regia. The toxicity of the heavy metals was evaluated using the national standard method; the

extraction procedure for toxicity of solid waste-horizontal vibration method (GB5086.2-1997). Sediment samples (2 g, air-dried, disaggregated and sieved to < 2 mm) were leached with acetic acid extraction solution at pH 5.0, 1:10 solid/liquid, and a rotational speed of 30 r/min for 8 h. The samples were not disturbed for 12 h, and they were finally filtered and analysed by atomic absorption spectrophotometry (AAS) (Z-5000, Hitachi, Japan). The specific resistance to filtration (SRF) of the treated sediment is tested with SRF equipment with settings of 34.5 kPa (pressure of vacuum pump) and 38.47 cm<sup>2</sup> (filtration area). The concentration of heavy metals in leachate (both dewatering and rainwater leachate) was measured with atomic absorption spectrophotometry (Z-5000, Hitachi, Japan).

All samples were processed in triplicates and the results are expressed as mean  $\pm$  standard deviations (S.D.). The data were analysed using an Analysis of Variance (SPSS 16.0) and a *t*-test to find the statistical significance ( $P < 0.01$ ).

## 2 Results and discussion

### 2.1 *In-situ* reduction of heavy metal toxicity

In this work, sediment aerating was used to suspend the sediments in the overlying water, in which the sulphide was oxidised. The results are presented in Table 2. After 30 d, over 99% of the AVS in the sediment was removed. ORP increased from -230 mV to well above 200 mV, and the sulphide odours disappeared. The results suggest that oxygen is an effective oxidant for sulphide. Other studies (Jia *et al.*, 1999; Caille *et al.*, 2003) have confirmed that during aerating, sulphides can be oxidised by oxygen and changed into sulphates, with the ORP values rapidly increasing.

The results of the extraction tests on the stabilisation-

**Table 2** Measurements of AVS and ORP in sediments

	AVS (mg/kg)	ORP (mV)
Initial sediment	3510 $\pm$ 214	-230 $\pm$ 27
Treated sediment	32.3 $\pm$ 5.0*	220 $\pm$ 25*

AVS: acid volatile sulphide; ORP: oxidation reduction potential.

\* Significance at  $P < 0.01$  compared with the control group, initial sediment.

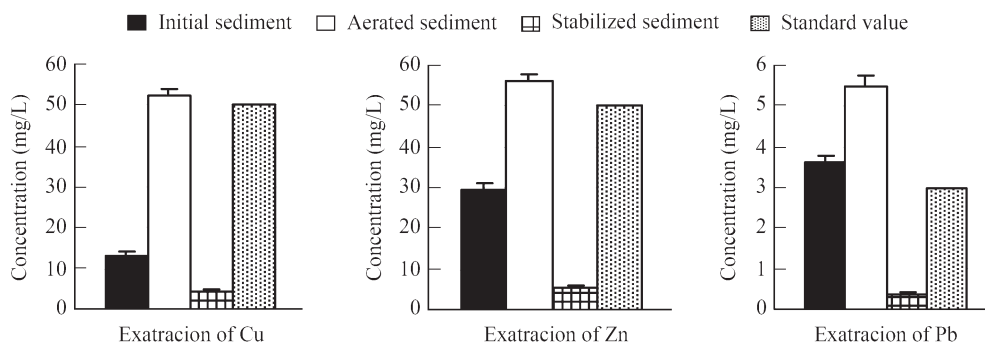
treated sediments are presented in Fig. 2. With exception of Pb (3.63 mg/L), the extraction concentrations the heavy metals (Cu and Zn at 13.04 and 29.54 mg/L, respectively) in the initial sediments are lower than the standard given in the Identification Standard for Hazardous Waste (GB5085.3-1996), which lists Cu, Zn and Pb at 50.0, 50.0 and 3.0 mg/L, respectively. However, after aerating the samples, the concentrations of the three metals increased to 52.3, 56.2 and 5.5 mg/L, values that all exceed the standard, but decreased sharply to 4.56, 5.25 and 0.35 mg/L after stabilisation treatment. The results show that the stabilisation efficiencies of Cu, Zn and Pb were about 65.0%, 82.2% and 90.0% compared to the initial sediments, and they were 91.3%, 90.7% and 93.6% compared to the aerating treatment sediment.

The chemical fraction analysis (Table 3) indicates that in the initial sediments Cu, Zn, and Pb mainly assume an organic-sulphuric form (42.2%, 33.9% and 33.2%) and a residual form. This may be a result of the large amount of the sulphide and organic material in the sediments. How-

**Table 3** Chemical fractions of Cu, Zn and Pb during *in-situ* sediment treatment

Sediment	Initial	Aerated	Stabilized
<b>Cu</b>			
Exchangeable (%)	1.3 $\pm$ 0.2	12.7 $\pm$ 1.3*	0.6 $\pm$ 0.1* <sup>a</sup>
Carbonate (%)	7.6 $\pm$ 0.3	4.8 $\pm$ 0.4*	2.7 $\pm$ 0.3* <sup>a</sup>
Fe-Mn oxide (%)	1.1 $\pm$ 0.2	6.6 $\pm$ 0.4*	4.7 $\pm$ 0.4* <sup>b</sup>
Organic (%)	42.2 $\pm$ 2.1	28.5 $\pm$ 1.8*	41.6 $\pm$ 2.1 <sup>a</sup>
Residual (%)	47.8 $\pm$ 3.3	47.5 $\pm$ 2.2	50.5 $\pm$ 2.0
Total conc. (mg/kg)	2418 $\pm$ 237.5	2604 $\pm$ 55.2	2055 $\pm$ 198.6 <sup>b</sup>
<b>Zn</b>			
Exchangeable (%)	9.6 $\pm$ 0.5	16.2 $\pm$ 1.4*	3.1 $\pm$ 0.3* <sup>a</sup>
Carbonate (%)	11.5 $\pm$ 0.6	10.6 $\pm$ 0.6	4.2 $\pm$ 0.3* <sup>a</sup>
Fe-Mn oxide (%)	4.8 $\pm$ 0.2	9.3 $\pm$ 0.4*	10.6 $\pm$ 0.6* <sup>b</sup>
Organic (%)	33.9 $\pm$ 1.8	21.6 $\pm$ 1.2*	38.7 $\pm$ 1.6* <sup>a</sup>
Residual (%)	40.3 $\pm$ 3.1	42.3 $\pm$ 2.1	43.4 $\pm$ 2.2
Total conc. (mg/kg)	2074 $\pm$ 205.3	2385 $\pm$ 226.7	1783 $\pm$ 169.4
<b>Pb</b>			
Exchangeable (%)	3.1 $\pm$ 0.2	8.2 $\pm$ 0.3*	1.7 $\pm$ 0.1* <sup>a</sup>
Carbonate (%)	6.3 $\pm$ 0.4	5.2 $\pm$ 0.2* <sup>a</sup>	3.2 $\pm$ 0.2* <sup>a</sup>
Fe-Mn oxide (%)	2.8 $\pm$ 0.2	6.2 $\pm$ 0.3*	8.4 $\pm$ 0.4* <sup>a</sup>
Organic (%)	33.2 $\pm$ 2.1	25.1 $\pm$ 1.9**	30.5 $\pm$ 2.0 <sup>b</sup>
Residual (%)	54.6 $\pm$ 3.0	55.3 $\pm$ 2.4	56.2 $\pm$ 2.3
Total conc. (mg/kg)	150.4 $\pm$ 12.4	171.4 $\pm$ 16.54	126.3 $\pm$ 11.9

\*, \*\* Significance at  $P < 0.01$ ,  $P < 0.05$  compared with initial sediment, respectively; <sup>a, b</sup> significance at  $P < 0.01$ ,  $P < 0.05$  compared with aerated sediment, respectively; no label indicates that there is no significant difference ( $P > 0.05$ ).



**Fig. 2** Extraction concentrations of Cu, Zn and Pb.

ever, as the sulphides oxidised, the bonded heavy metals became easy to release. As a result, the concentrations of the three metals in the organic-sulphuric fractions reduced by 8.1%–13.7%, while the fractions of the exchangeable forms increased by 5.2%–11.3%. This is the main reason why the extraction concentrations of Cu, Zn and Pb in sediments treated by aerating exceed the standard. Thus, it can be concluded that when the sediment was transformed from reduced to oxidised state, the sulphuric fractions of the heavy metals were not stable. In other words, the sediments with a serious sulphide odour problem are potentially toxic, although the extraction concentrations of the heavy metals in the initial sediments were low.

For the stabilisation treated sediments, compared to the aeration treated sediments, the total concentrations of Cu, Zn, and Pb in exchangeable carbonate forms decreased by 14.2%, 19.5%, and 8.4%, while the organic form increased by 13.2%, 17.2%, and 5.4%, respectively. This suggests that in the absence of sulphide, the increase in the amount of heavy metals concentrated in organic forms after stabilisation treatment may be due to the formation of heavy metal phosphates. Previous studies on the combination of heavy metal and phosphate (Cotter *et al.*, 1996) have found that the formation of heavy metal phosphate is effective in reducing the solubility or bioavailability of heavy metals, especially the phosphate-Pb composite, which was considered to be stable over a wide range of environmental conditions. The exact reaction mechanisms that provide for binding between heavy metals and phosphates (when coexisting with other chemicals, such as lime) need further study. From the results of the chemical fractions analysis, we hypothesize that the stabilisation of heavy metals by the stabilisation reagents (phosphates and lime) is caused by the formation of organic and residual fractions.

## 2.2 Dewater treatment of dredged sediment

The sediments were dredged and mechanically dewatered as non-hazardous waste, which was followed by *in-situ* pretreatment. SRF of the treated sediments, the water content of mud cakes after mechanical dewatering and the concentrations of Cu, Zn and Pb in leachate are shown in Table 4.

It is generally accepted that sediment dewatering capability is very poor when  $\text{SRF} > 1.0 \times 10^9 \text{ s}^2/\text{g}$ , moderate when  $0.5 \times 10^9 \text{ s}^2/\text{g} < \text{SRF} < 0.9 \times 10^9 \text{ s}^2/\text{g}$ , and high when  $\text{SRF} < 0.5 \times 10^9 \text{ s}^2/\text{g}$ . Table 4 shows that the dewatering capability of the original sediment is quite poor with

**Table 4** Characteristic of the sediments before and after dewatering

	Initial sediment	Treated sediment
SRF ( $\text{s}^2/\text{g}$ )	$(1.13 \pm 0.22) \times 10^9$	$(0.12 \pm 0.03) \times 10^9$ *
Water content (%)	85 $\pm$ 5	65 $\pm$ 3**
Concentration in the leachate (mg/L)		
Cu	2.256 $\pm$ 0.134	0.529 $\pm$ 0.059*
Zn	2.150 $\pm$ 0.126	0.893 $\pm$ 0.084*
Pb	0.689 $\pm$ 0.061	0.076 $\pm$ 0.015*

\*, \*\* Significance at  $P < 0.01$ ,  $P < 0.05$ , respectively, compared to initial sediment.

SRF: specific resistance to filtration.

SRF measurement, which was  $1.13 \times 10^9 \text{ s}^2/\text{g}$ . Using stabilisation treatment (injecting lime and PAM), the dewatering capability improved and SRF was  $0.12 \times 10^9 \text{ s}^2/\text{g}$ . The water content of the mud cake dropped from 85% in the initial sediment to 65% in the treated sediment, which improved the HTAC treatment of the sediment. The technique of using both lime and PAM to improve the sediment dewatering capability via a flocculate reaction has been reported by Lee and liu (2000).

The concentrations of Cu, Zn and Pb in the dewatering leachate of the treated sediments dropped by 60%–89% in comparison with the initial sediment. These values meet the standards in EQSSW(GB3838-2002), which are 1.0, 2.0 and 0.1 mg/L, respectively. As a result, the leachate can be directly discharged into the river without further treatment.

## 2.3 Chemical fractions of heavy metals during the composting

During HTAC treatment, the temperature steadily increased to 72°C on day 3 and the pH fluctuated between 8.0 and 8.5. The treatment was accomplished with a cycle about 20 d. The final compost product was pitchy and the physiochemical characteristics are shown in Table 5. After HTAC treatment, the extraction concentrations of Cu, Zn and Pb are much lower than the national standard values given in the Identification Standard for Hazardous Waste. This result indicates that the heavy metal toxicity was further reduced by composting treatment.

**Table 5** Physiochemical characteristics of the compost product

	Mean	Extraction concentration (mg/L)
Organic matter (g/kg)	78.44	
TN (g/kg)	2.17	
TP (g/kg)	1.28	
pH	8.2	
Heavy metal		
Cu (mg/kg)	2680	4.42 $\pm$ 0.32
Zn (mg/kg)	2298	5.03 $\pm$ 0.46
Pb (mg/kg)	162.6	0.25 $\pm$ 0.11

The chemical fractions of Cu, Zn and Pb were analyzed during HTAC and the results are presented in Table 6. It is worth notice that there is an increase (about 20%) in the total concentrations of the three kinds of heavy metals after HTAC treatment. This may be due to the decrease in the composting volume, which can be caused by the dissipation of water and the decomposition and volatilisation of organic matter during HTAC treatment (Zheng *et al.*, 2005). However, as mentioned above, the ecological impact of heavy metals in the soil environment is more related to their chemical forms rather than their total concentrations (Hsu and Lo, 2001; Nomeda *et al.*, 2008; He *et al.*, 2008). Generally, heavy metals in an exchangeable form, a carbonate form, and an Fe-Mn oxide form are considered to be highly transportable and highly bio-available. As shown in Table 6, the content percentage of these three forms all decreased, while the organic and residual forms increased, which indicates that the heavy

**Table 6** Chemical fractions of Cu, Zn and Pb before and after composting

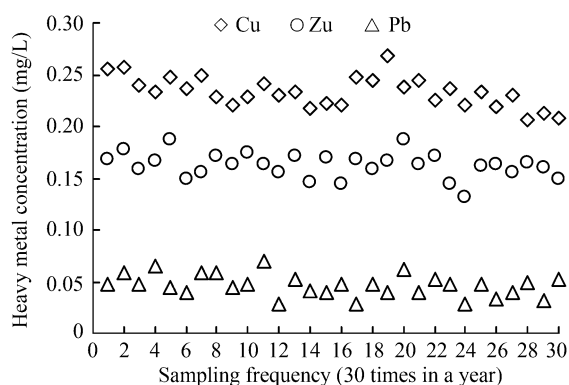
Chemical fraction		Exchangeable (%)	Carbonate (%)	Fe-Mn oxide (%)	Organic (%)	Residual (%)	Total concentration (mg/kg)
Cu	Before	0.6 ± 0.1	2.7 ± 0.3	4.7 ± 0.4	41.6 ± 2.1	50.5 ± 2.0	2055 ± 198.6
	After	0.5 ± 0.1	1.6 ± 0.2**	3.5 ± 0.3**	42.8 ± 2.2	51.7 ± 3.1	2562 ± 243.4**
Zn	Before	3.1 ± 0.3	4.2 ± 0.3	10.6 ± 0.6	38.7 ± 1.6	43.4 ± 2.2	2385 ± 226.7
	After	1.3 ± 0.2*	2.3 ± 0.3*	5.3 ± 0.3*	46.3 ± 2.4**	44.9 ± 3.3	2863 ± 256.8**
Pb	Before	1.7 ± 0.1	3.2 ± 0.1	8.4 ± 0.4	30.5 ± 2.0	56.2 ± 2.3	171.4 ± 16.5
	After	1.1 ± 0.1*	1.7 ± 0.3*	5.1 ± 0.6*	34.6 ± 2.4	57.5 ± 3.6	234.3 ± 24.6*

\*, \*\* Significance at  $P < 0.01$  and  $P < 0.05$  compared with group of before treatment, respectively; no label indicates that there is no significant difference ( $P > 0.05$ ).

metals in the sediments had been stabilised during the HTAC treatment. This finding largely agrees with previous reports (Canarutto *et al.*, 1991; Greenway and Song, 2002).

## 2.4 Monitoring results of heavy metals concentrations in rainwater leachate

Compost products were added directly to the soil test area and rainwater leachate was collected on rainy days. In a year, the concentrations of Cu, Zn and Pb (Fig. 3) in rainwater leachate stayed below 0.2, 0.3 and 0.1 mg/L, respectively. Those values are much lower than the standards in EQSSW (GB3838-2002). These results indicate that the toxicity of the heavy metals in sediments was markedly reduced due to the *in-situ* stabilisation followed by an *ex-situ* composting process.



**Fig. 3** Monitoring results of heavy metal concentrations in rainwater leachate.

## 3 Conclusions

In summary, using this new process of *in-situ* stabilisation followed by *ex-situ* composting treatment, the toxicity of heavy metals polluted sediments (such as hazardous material) was markedly reduced, and the sediments could be treated and disposed as non-hazardous material. The end product can then be used as fertiliser for the river bank plants (in this case).

After *in-situ* stabilisation, the extraction concentrations of Cu, Zn and Pb decreased from 13.04, 29.54, and 3.6 mg/L to 4.56, 5.25, and 0.35 mg/L, respectively. These final values are much lower than the national standards of ISHW (GB5085.3-1996).

The dewatering capability of the treated sediment increased, while the water content of the dewatered sediment decreased from 85% to 65%, which improved the fol-

lowing HTAC of the treated sediment. Chemical fractions analysis showed that Cu, Zn and Pb were further stabilised during the HTAC process.

Yearly monitoring of the heavy metal concentrations in rainwater leachate through the bank testing area showed that the concentrations of these three heavy metals meet the values of V standard of the EQSSW (GB3838-2002).

## Acknowledgments

This work was supported by the Scientific and Technological Planning Project of Guangdong Province (No. 2003A3040404) and the Guangdong & Hong Kong Technology Cooperation Funding (No. 2006A36702001). The authors acknowledge the Government of Guangzhou City for providing the application locale for this study. We also thank all of the members of the project team for their valuable insights and assistance with this work.

## References

- Aparna C, Saritha P, Himabindu V, Anjaneyulu Y, 2008. Techniques for the evaluation of maturity for composts of industrially contaminated lake sediments. *Waste Management*, 28: 1773–1784.
- Brouwer H, Murphy T P, 1994. Diffusion method for the determination of acid volatile sulphide in sediment. *Environmental Toxicology and Chemistry*, 13: 1273–1275.
- Caille N, Tiffreau C, Leyval C, Morel J L, 2003. Solubility of metals in an anoxic sediment during prolonged aeration. *The Science of the Total Environment*, 301: 239–250.
- Canarutto S, Petruzzelli G, Lubrano L, Guidi G V, 1991. How composting affects heavy metal content. *Biocycle*, 32(6): 48–50.
- Chowdhury A K, Stolzenburg T R, Stanforth R R, Warner M A, LaRowe M E, 1996. Underwater treatment of lead-contaminated sediment. *Remediation Journal*, 6(2):15–21.
- Cotter H J, Caporn S, 1996. Remediation of contaminated land by formation of heavy metal phosphates. *Applied Geochemistry*, 11: 335–342.
- Fang T, Xiao B D, Zhang X H, Ao H Y, Xu X Q, 2002. Effect of aeration on heavy metals release from two different sediments. *China Environmental Science*, 22(4): 355–359.
- Greenway G M, Song Q, 2002. Heavy metal speciation in the composting process. *Journal of Environmental Monitoring*, 4: 300–305.
- Gillian M G, Qi J S, 2002. Heavy metal speciation in the composting process. *Journal of Environmental Monitoring*, 4: 300–305.
- He M M, Tian G M, Liang X Q, 2009. Phytotoxicity and speciation of copper, zinc and lead during the aerobic composting of sewage sludge. *Journal of Hazardous Materials*, 163(2):

- 671–677.
- Hettiarachchi G M, Pierzynski G M, Ransom M D, 2001. *In situ* stabilization of soil lead using phosphorus. *Journal of Environmental Quality*, 30: 1214–1221.
- Hsu J H, Lo S L, 2001. Effect of composting on characterization and leaching of copper, manganese, and zinc from swine manure. *Environmental Pollution*, 114: 119–127.
- Jia Z B, Lam K CH, Liu F W, 1999. Effect of aeration of Hong Kong sediment on lead binding. *Acta Scientiarum Naturalium Universitatis Pekinensis*, 35(6): 834–841.
- John C S, Jessica M H, Brian P J, Vijay M V, 2003. *In situ* treatment of metals in contaminated soils with phytate. *Journal of Environmental Quality*, 32: 153–161.
- Jon R, 1998. *In situ* treatment of contaminated sediments. Technology status report prepared for the U.S. EPA Technology Innovation Office. Washington, DC. 3–6.
- Lee C H, Liu J C, 2000. Enhanced sludge dewatering by dual polyelectrolyte conditioning. *Water Research*, 34(18): 4430–4436.
- Li M G, Zhong J H, Li S, Zheng W, 2005. Investigation and assessment of sediment pollutants in the brooklets in Guangzhou. *Guangzhou Environmental Science*, 20(4): 1–5.
- Nomeda S, Valdas P, Chen S Y, Lin J G, 2008. Variations of metal distribution in sewage sludge composting. *Waste Management*, 28: 1637–1644.
- Ottaviani M, Veschetti E, Petruzzelli G, Lubrano L, 1994. Characterization of heavy metal mobile species in sewage sludge for agricultural utilisation. *Agrochimica*, 38: 277–284.
- Tessier A, Campbell P G C, Bisson M, 1979. Sequential extraction procedure for the speciation of particulate trace metals. *Analytical Chemistry*, 51(7): 844–851.
- Wang Y M, Chen T C, Yeh K J, Shue M F, 2001. Stabilization of an elevated heavy metal contaminated site. *Journal of Hazardous Materials*, 88(1): 63–74.
- Wei Y J, Liu Y S, 2005. Effects of sewage sludge compost application on crops and cropland in a 3-year field study. *Chemosphere*, 59: 1257–1265.
- Zheng G D, Chen T B, Gao D, Luo W, Li Y X, 2005. Influence of high temperature aerobic composting treatment on the form of heavy metals in pig manure. *China Environmental Science*, 25(1): 6–9.
- Zhuang Y Y, Allen H E, Fu G, 1994. Effect of aeration of sediment on cadmium binding. *Environmental Toxicology and Chemistry*, 13(5): 717–724.