



Migration of Cu, Zn and Cr through municipal solid waste incinerator bottom ash layer in the simulated landfill



Jun Yao^{a,b}, Zhanhong Qiu^a, Qingna Kong^a, Luxi Chen^a, Huayue Zhu^a, Yuyang Long^c, Dongsheng Shen^{c,*}

^a Centre of Marine and Ecological Environment Protection, Taizhou University, Linhai 317000, China

^b Department of Environmental Engineering, Zhejiang University, Hangzhou 310029, China

^c Zhejiang Provincial Key Laboratory of Solid Waste Treatment and Recycling, Zhejiang Gongshang University, Hangzhou 310018, China

ARTICLE INFO

Article history:

Received 16 December 2016

Received in revised form 26 February 2017

Accepted 27 February 2017

Available online 8 March 2017

Keywords:

MSWI bottom ash layer

Cu

Zn

Cr

Migration

ABSTRACT

Municipal solid waste incinerator (MSWI) bottom ash is used as the cover, intermediate cover and liner in the landfill. In this study, simulated landfills were operated for 507 days to investigate the effect of MSWI bottom ash layer on the migration of Cu, Zn and Cr. All through the study, Cu was greatly released from the MSWI bottom ash layer. Zn was captured by the MSWI bottom ash layer at the initial stage of the landfill. After then, the release of Zn from the MSWI bottom ash layer was observed, corresponding to the decline of the leachate pH. No trapping or release was found for Cr at the initial stage of the landfill. Then, a slight release of Cr from the MSWI bottom ash layer was observed. Over the study, a totally 0.93% of Cu, 0.12% of Zn, and 0.01% of Cr in the MSWI bottom ash layer were released. The released metals were not discharged to the surrounding environment. They were immobilized by the sub-MSW layer. It suggested that the MSWI bottom ash layer could not increase the metal discharge when it was used as the cover or intermediate cover in the landfill.

© 2017 Elsevier B.V. All rights reserved.

1. Introduction

Incineration has become more and more important for the treatment of municipal solid waste (MSW) due to the shortage of the urban land as well as the advantages of hygienic control, volume and mass reduction and energy recovery. For instance, in the past ten years, the number of municipal solid waste incinerator (MSWI) facilities increased sharply from 47 to 166 in China, with the treatment capacity rising from 15,000,000 to 158,488,000 kg/d (National Bureau of Statistics of China, 2015). In the incineration process, about 20% of the incinerated MSW was transformed to the solid residue. Among the residue, bottom ash is the main stream, which accounts for approximately 80% (Chimenos et al., 1999). MSWI bottom ash is allowed to be disposed in the MSW landfill sites in several countries and areas, such as China, Japan and Taiwan (Inanc et al., 2007; Lo and Liao, 2007; Youcai et al., 2002). The compacted ash can have a hydraulic conductivity similar to that of clay, which means that it could meet the functional requirements as the cover and liner in the landfill (Chandler et al., 1997; Muhunthan et al., 2004). Therefore, it has been used as the cover, intermediate cover and

liner instead of natural clay minerals in the landfill (Su et al., 2013). For example, Travar et al. (2009) used the MSWI ash as the substitute for natural materials in landfill cover construction. Li et al. (2014) used the MSWI bottom ash as the intermediate layer. These studies suggested that the MSWI bottom ash layer could provide several advantages for the landfill construction and operation. For example, it could save the natural mineral material source, fasten the stabilization of the landfill and improve the leachate quality.

MSWI bottom ash is a mineral assemblage, containing a high level of alkaline minerals, adsorption medium and heavy metals (Yao et al., 2010; Yao et al., 2015). The alkaline minerals in the MSWI bottom ash result in the high acid neutralization capacity (ANC), which can increase the pH of the leachate vertically flowing through the MSWI bottom ash layer. As the mobility of the heavy metals is dependent on the pH, the migration of them can be changed. The adsorptive medium, including Friedel's salt, kaolin and iron (hydr)-oxides, were reported to have adsorption capacity for the metals. For instance, Dai et al. (2009) pointed out that Friedel's salt could remove 99% of CrO_4^{2-} from the aqueous solution. Adebowale et al. (2005) suggested that kaolin showed a strong affinity for Cu, Zn, Pb and Cd. The ANC and adsorptive medium can help the MSWI bottom ash layer capture the heavy metals from the leachate. However, on the other hand, the high content of heavy metals in the MSWI bottom ash can lead to the heavy metal release, which may aggravate

* Corresponding author.

E-mail address: yizaghi@126.com (D. Shen).

the heavy metal pollution (Yao et al., 2010). Several studies have been done to investigate the effect of the disposal of MSWI bottom ash on the heavy metal leaching from the landfill (Inanc et al., 2007; Lo and Liao, 2007; Su et al., 2013). In these studies, MSWI bottom ash was blended with the MSW. The effect of the co-disposal on the final discharge of heavy metals from the landfill was discussed. However, the effect of MSWI bottom ash layer on the migration of heavy metals in the landfill is still poorly investigated. This theme is important as it can provide scientific reference for the control of heavy metal pollution when the MSWI bottom ash layer is used as the cover, intermediate liner and liner in the landfill.

Among the heavy metals, Cu and Zn are presented in high concentration in MSWI bottom ash and MSW (Long et al., 2010; Yao et al., 2014). Cr, as a valence variable metal, is of particular concern as its high toxicity to the surrounding ecosystem (Paquin et al., 2000). They were thus selected for the discussion herein.

The main aim of this study is to reveal the effect of the MSWI bottom ash layer on the migration of heavy metals in the landfill. Two parallel leachate recirculated landfill bioreactors were established and operated for 507 days. MSWI bottom ash was disposed as the landfill layer. The variation of pH, chemical oxygen demand (COD), Cu, Zn and Cr concentrations in the leachates, which were sampled above the MSWI bottom ash layer (L1), beneath the MSWI bottom ash layer (L2) and at the bottom of the landfill (L3), was monitored.

2. Material and methods

2.1. Experimental setup

Two parallel simulated landfill bioreactors with leachate recirculation were set up. The reactor was 287 mm in diameter and 1430 mm in height, with a total working volume of 92 L. Five ports were equipped with the reactor: the two ports at the top were used for gas exporting and leachate recirculation; the two ports at the side were used for sampling the leachate above (L1) and beneath (L2) the MSWI bottom ash layer; the port at the bottom was used for leachate drainage and sampling (L3). A 100 mm thick layer of gravel was placed at the bottom of the landfill to simulate a leachate collection system and to prevent clogging of the leachate withdrawal outlets. The MSWI bottom ash layer was placed between the MSW layers. Another 50 mm thick layer of sand was placed at the top of each landfill to simulate the cover and top drainage layer. The schematic of the simulated landfill system was shown in Fig. 1.

2.2. MSWI bottom ash and MSW

MSWI bottom ash was sampled from the Green Energy MSWI plant in Zhejiang province, East China. The plant consists of two parallel stoker incinerators with an MSW treatment capacity of 650,000 kg/d. The MSWI bottom ash sample had undergone water quenching and magnetic separation before being sampled. A part of the MSWI bottom ash sample was mingled, air dried and ground into less than 154 μm for total heavy metal analysis and sequential extraction procedure (SEP).

MSW was collected from the Jiazhi transport station of Taizhou, Zhejiang, East China. It was sampled in the morning, afternoon and night on the same day. Then, the large particle of the refuse was shredded into approximately 20 mm. The shredded refuse was homogenized by a shovel as thoroughly as possible before loaded to the landfill reactor. The moisture content of the refuse was 59.6%. The components of the MSW are shown in Table 1.

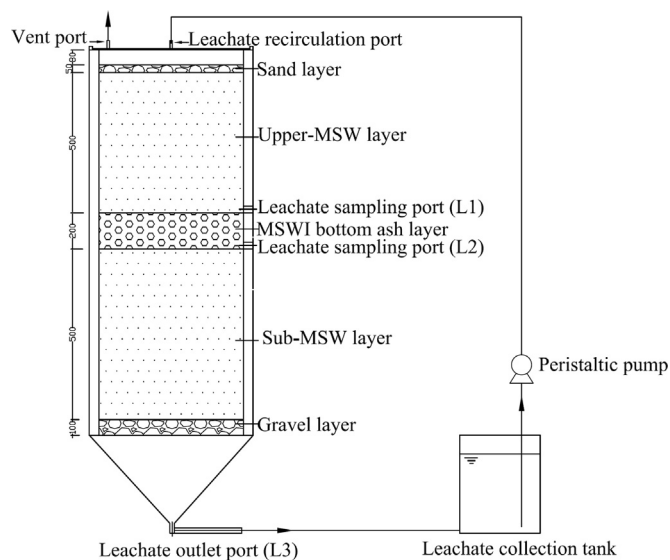


Fig. 1. Schematic of the simulated landfill system.

2.3. Operation of the reactors

Firstly, about 25 kg MSW was loaded to the each reactor and compacted using the shovel and sledgehammer. Then, 16 kg MSWI bottom ash was loaded and compacted. Finally, another 25 kg MSW was loaded and compacted. The mass proportion of MSWI bottom ash to MSW in Zhejiang province, China. The moisture of the MSW was adjusted to 75% by adding tap water, which was reported to be an initial rapid decomposition threshold for the anaerobic organic refuse mineralization in bioreactor landfill (Benson et al., 2007; Lay et al., 1998). After the loading, the reactors were sealed with a gasket and silicone sealant. The leachates generated from the landfill reactors were collected in the leachate collection tank and continuously recirculated using pumps with adjusted flow rates varying with the leachate volume. To keep the volume equilibrium of the leachate, a same volume of tap water was added to the reactors after the sampling.

2.4. Total heavy metals analysis

MSWI bottom ash was digested using the method described in the previous study (Yao et al., 2010). After the digestion, Cu, Zn and Cr concentrations in the solution were determined by an Inductively Coupled Plasma-Atomic Emission Spectroscopy (ICP-AES).

Chemical speciation of Cu, Zn and Cr in the bottom ash sample was determined using SEP suggested by Tessier et al. (1979). The procedure classified elements into five fractions: (1) exchangeable fraction (F1), (2) carbonate bound fraction (F2), (3) Fe-Mn oxide bound fraction (F3), (4) organic matter bound fraction (F4), and (5) residual fraction (F5).

2.5. Analytical procedure

L1, L2 and L3 were collected weekly from the landfill reactors. The leachate samples were analyzed for pH, chemical oxygen demand (COD), concentrations of Cu, Zn and Cr. The analyses of pH and COD were performed in accordance with Chinese standard methods GB 6920-86 and GB 11914-89, respectively. The total concentrations of Cu and Zn and Cr were determined by ICP-AES. Cr(VI) were determined by Chinese standard methods GB 7467-87. The results were expressed as the mean of the obtained data from the two reactors.

Table 1
Components of the MSW

Components	Food waste	Plastic	Paper	Textile	Dust	Ceramic	Metal	Timber	Residue
W/W, %	44.3	8.2	7.5	0.3	6.2	5.1	0.1	1.7	27.6

Table 2
Physicochemical properties and bulk composition of the MSWI bottom ash sample

Properties or elements	Value
Physi-chemical properties	
Moisture content (%)	1.61
Bulk density (kg/m ³)	1277.6
Loss on ignition (LOI) (%)	2.2
pH	11.2
Acid neutralization capacity (ANC _{7.5})	1.0H ⁺ mmol g ⁻¹
Element composition (mg kg⁻¹)	
Al	40920 ± 1600
Si	223600 ± 4657
Na	9040 ± 178
K	15792 ± 167
Mg	5997 ± 115
Ca	69413 ± 2613
Fe	26008 ± 28
Mn	1246 ± 231
Zn	1922 ± 33
Cu	315 ± 22
Cr	252 ± 42
Cd	7 ± 1
Mo	7 ± 2
As	138 ± 42
Co	21 ± 2
Ni	48 ± 24

Table 3
Fractionation distribution of Cu, Zn and Cr in MSWI bottom ash (mg kg⁻¹)

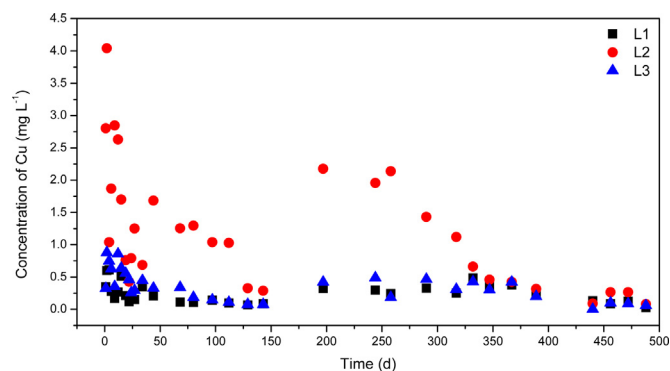
	F1	F2	F3	F4	F5
Cu	3.73	74.61	7.86	141.60	86.79
Zn	0.96	452.55	485.95	79.01	903.53
Cr	3.41	8.67	25.44	4.76	209.42

3. Result and discussion

3.1. Main characteristic of MSWI bottom ash

The result of physicochemical properties and bulk composition analysis is shown in Table 2. The alkali metals, including Na, K, Al, Ca and Mg, were abundant in the MSWI bottom ash sample, which contributed to the high pH (11.2) and acid neutralization capacity (1.0H⁺ mmol g⁻¹). Cu, Zn and Cr were the major trace elements. Their contents in the MSWI bottom ash were 4.1, 14.9 and 25.9 times higher than those in the soils (National Environmental Monitoring Centre of China, 1990), indicating a high potential environmental risk.

The chemical speciation of Cu, Zn and Cr is shown in Table 3. Most of Cu, Zn and Cr were occupied as the organic matter bound fraction (F4) and residual fraction (F5), which were considered as the stable fractions. However, there were still certain amounts of Cu, Zn and Cr occupied as the exchangeable fraction (F1), carbonate bound fraction (F2) and Fe-Mn oxide bound fraction (F3), which were thought to be unstable with the potential leachability. The total amounts of the unstable Cu, Zn and Cr in the MSWI bottom ash were 86.21, 939.46 and 37.52 mg kg⁻¹, respectively. The unstable Cr was less than the unstable Cu and Zn.

**Fig. 2.** Variation of Cu concentrations of L1, L2 and L3.

3.2. Migration of Cu, Zn and Cr through the MSWI bottom ash layer

3.2.1. Cu

Generally, the concentrations of Cu in L1, L2 and L3 showed a decreasing trend (Fig. 2), which was consistent with the previous report (Long et al., 2009). The concentration of Cu in the leachate was relatively high at the beginning of the study, which was due to the intensive release of the unstable Cu from the MSW and MSWI bottom ash. Our previous study had showed that the sulfate in the landfill was reduced to sulfide with the time extension, leading to the increase of sulfide level (Long et al., 2010). S²⁻ could react with Cu²⁺ to form CuS (Eq. (1)), a compound of copper with a low solubility. Therefore, the Cu concentration of the leachate was attenuated.



The Cu concentration of L2 was higher than that of L1 all through the study. The average Cu concentration of L2 was 1.22 mg L⁻¹, while the average Cu concentration of L1 was 0.25 mg/L. It meant that Cu was largely released from the MSWI bottom ash layer. Due to the high acid neutralization capacity (ANC) of MSWI bottom ash (Table 2), the pH of the leachate increased after flowing through the MSWI bottom ash layer (Fig. 3(a)). Notably, the pH of L1 ranged from 4.27 to 6.13 in the first 42 days, while the pH of L2 ranged from 9.07 to 11.76. The increase of pH was thought to be able to immobilize the metal and decrease the Cu concentration of the leachate. However, the Cu concentration of the leachate increased after it flowed through the MSWI bottom ash layer. Cu is reported to have a good affinity with the organic matter. Several researches have showed that the leaching of Cu from MSWI bottom ash could be greatly enhanced by the organic matter (Meima et al., 1999; Olsson et al., 2007; van Zomeren and Comans, 2004). Landfill leachate contained a high level of organic matter. The COD of the leachate exceeded 10000 mg/L all through the study (Fig. 3(b)). The release of Cu from the MSWI bottom ash layer was probably facilitated by the organic matter in the leachate. To approve the assumption, the distribution pattern of Cu in L2 was analyzed by the model Visual MINTEQ. It showed that Cu in L2 was almost 100% bound with the organic matter (Table 4). This result verified the role of organic matter on the Cu release from the MSWI bottom ash layer. It should be noted that the difference of Cu concentrations in L2 and L1 declined with the extension of time. After day 335, the Cu concentration of L1 became close to that of L2, suggesting the release of Cu receded. According

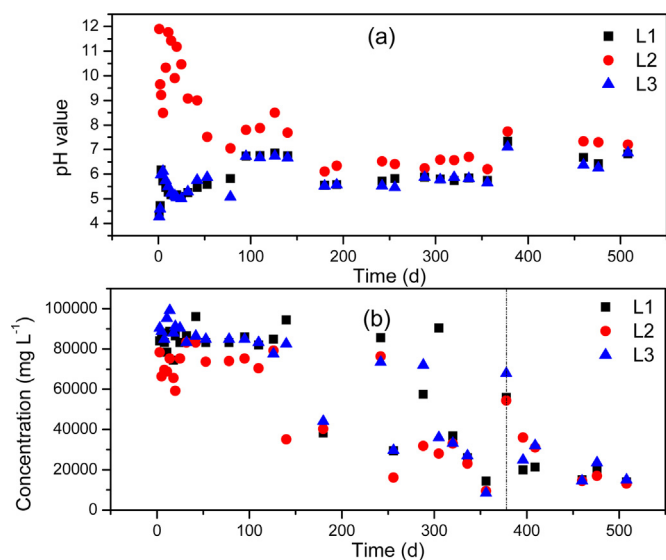


Fig. 3. Variation of pH (a) and COD (b) of L1, L2 and L3.

Table 4
Speciation of Cu in L2 calculated by Visual MINTEQ (mol L^{-1})

Speciation	Day 3	Day 193	Day 456
Bound to organic matter	1.64×10^{-5}	3.42×10^{-5}	4.15×10^{-6}
Bound to inorganic matter	3.78×10^{-18}	$1.07\text{E} \times 10^{-15}$	1.19×10^{-14}

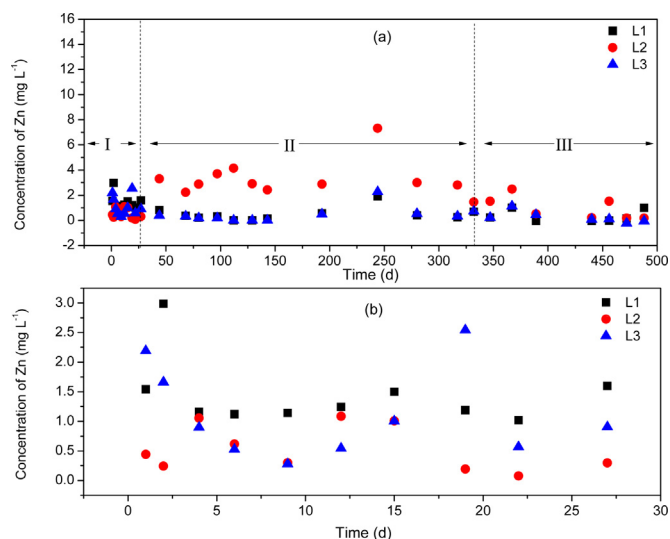


Fig. 4. Variation of Zn concentrations of L1, L2 and L3 over the study (a) and in the first 27 days (b).

to the difference of Cu concentrations in L1 and L2, as well as the volume of the leachate, it was calculated that 46.88 mg of Cu was released from the MSWI bottom ash layer during the study, which accounted for 0.93% of its total amount in the MSWI bottom ash.

3.2.2. Zn

The decreasing trend was also observed for the Zn concentrations of L1 and L3 (Fig. 4(a)), which could be attributed to the increase of pH and formation of insoluble minerals. For L2, the concentration of Zn was relatively low in the first 27 days (Fig. 4(b)), which was corresponding to the high pH. After day 27, the pH of L2 decreased to below 9.0, leading to the sharp increase of Zn concentration.

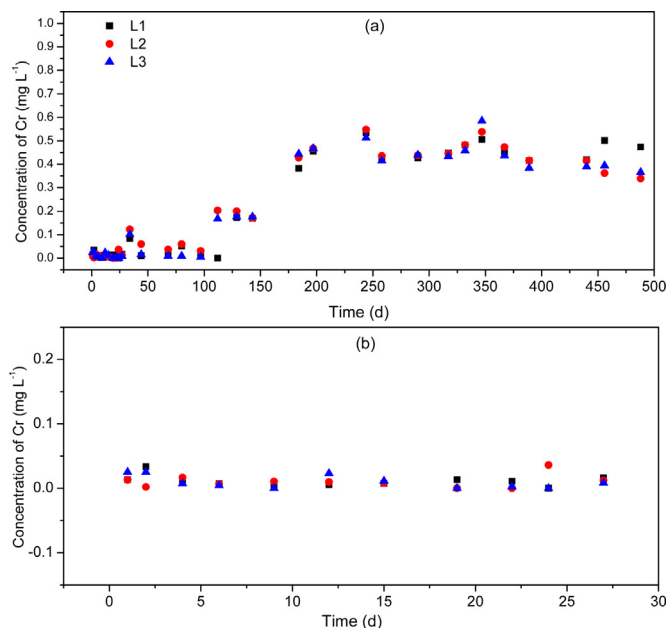


Fig. 5. Variation of Cr concentrations of L1, L2 and L3 over the study (a) and in the first 27 days (b).

According to the comparison of Zn concentrations of L1 and L2, the running of the landfill could be divided into three stages. At Stage I (day 1 to day 27), the Zn concentration of L2 was lower than that of L1, indicating the MSWI bottom ash layer captured Zn from the leachate. It might be partly resulted from the hydrolyzing of the Zn under the alkaline condition, which formed the insoluble compounds, such as $\text{Zn}(\text{OH})_2$ and ZnCO_3 . Besides, Zn could also be adsorbed by the adsorptive medium in the MSWI bottom ash. At Stage II (day 28 to day 335), the Zn concentration of L2 was higher than that of L1, indicating Zn was released from the MSWI bottom ash layer. At Stage III (day 336 to day 488), the Zn concentrations of L1 and L2 were close, suggesting the adsorption and release of Zn came to a balance. According to the leachate volume and difference of Zn concentrations of L1 and L2, a totally 13.74 mg of Zn was captured by the MSWI bottom ash layer at Stage I. However, 50.90 mg of Zn was released from the MSWI bottom ash layer at the Stage II. As a whole, 37.16 mg of Zn was released from the MSWI bottom ash layer, which accounted for 0.12% of its total amount in the MSWI bottom ash layer.

3.2.3. Cr

Different from Cu and Zn, the concentrations of Cr in L1, L2 and L3 showed an increasing trend (Fig. 5(a)). At the beginning of the study, the concentration of Cr was at a low level (Fig. 5(b)). It suggested that the initial release of Cr from the MSW and MSWI bottom ash was very limited. With the degradation of MSW, Cr was gradually released from the MSW, resulting in the increase of Cr concentration of the leachate.

The Cr concentration of L2 was close to that of L1 in the first 27 days (Fig. 5(b)). At the initial stage of the landfill, most of Cr in the leachate was presented as Cr(VI) (Table 5), which was probably due to the residual oxygen in the landfill. The adsorptive medium in the MSWI bottom ash, such as the Friedel's salt and kaolin, could capture the Cr(VI) from the leachate (Adebowale et al., 2005; Dai et al., 2009). On the other hand, MSWI bottom ash contained considerable amounts of leachable Cr (Table 2), which could be released during the study. The trapping and release of Cr seemed to be very close, resulting in the close concentrations of Cr in L1 and L2. From day 28 to day 357, as the pH of L2 decreased significantly, the adsorption of Cr(VI) was restricted. Besides, the decrease of the pH was bene-

Table 5
Proportion of Cr(VI) of the total chromium in the leachates (%)

Time	L1	L2	L3
Day 9	111.22	108.91	114.40
Day 129	84.58	64.34	36.31
Day 244	17.53	18.42	19.12
Day 347	25.88	25.11	19.33
Day 488	20.36	23.82	17.05

ficial to the release of Cr. Therefore, the release of Cr exceeded the adsorption. After day 357, the Cr concentration of L2 was lower than that of L1. At that time, most of Cr was presented as Cr(III) under the anaerobic condition (Table 5). As the pH of L2 was higher than that of L1, Cr(III) was immobilized by the MSWI bottom ash layer with the formation of Cr(OH)₃. According to the leachate volume and the difference of Cr concentrations of L1 and L2, it was calculated that 0.78 mg of Cr was released from the MSWI bottom ash layer from day 28 to day 357, while 0.42 mg of Cr was captured by the MSWI bottom ash layer after day 357. Thus, as a whole, 0.36 mg of Cr was released from the MSWI bottom ash layer. The released Cr was lower than that of Cu and Zn, which could be due to its low unstable amounts in the MSWI bottom ash as well as its variable valence state in the landfill.

3.3. Fate of the released metals and the implication for the design of the landfill

Above results showed that 46.88 mg of Cu, 37.16 mg of Zn, and 0.36 mg of Cr were released from the MSWI bottom ash layer. However, the concentrations of Cu, Zn and Cr in L3 were generally at the same level with those of L1. For example, the average Cr concentration of L1 was 0.205 mg/L, while the average Cr concentration of L3 was 0.203 mg/L. It suggests that the MSWI bottom ash layer can not increase the final discharge of heavy metals from the landfill. The released Cu, Zn and Cr were probably adsorbed by the sub-MSW layer. Our previous study showed that the MSWI bottom ash layer increased the Cu and Zn contents of the MSW in the bottom layer (Yao et al., 2014). Lo et al. (2009) pointed out that MSW had a great adsorption capacity for heavy metals. These results were in agreement with the result of this study. When the MSWI bottom ash layer is used as the cover or intermediate cover, the released heavy metals can be immobilized by the sub-MSW layer, which can prevent the increase of metal discharge from the landfill. When the MSWI bottom ash layer is used as the liner, the permeability coefficient must be strictly controlled to prevent the contamination of the underground water by the percolated leachate. As the liner has no sub-MSW layer served as the barrier for the released heavy metals like the cover or intermediate cover. Therefore, from an environmental view, the MSWI bottom ash layer should be preferentially used as the cover or intermediate cover in the landfill.

4. Conclusion

The MSWI bottom ash layer can affect the migration of Cu, Zn and Cr in the landfill. The concentration of Cu in the leachate increased after it flowed through the MSWI bottom ash layer, due to the intensive release of Cu from the MSWI bottom ash layer. The MSWI bottom ash layer captured the Zn in the leachate at the initial stage of the landfill. After then, the release of Zn from the MSWI bottom ash was observed. The MSWI bottom ash layer had little effect on the migration of Cr at the initial stage of landfill. Afterwards, the release of Cr from the MSWI bottom ash layer resulted in the increase of Cr concentration of the leachate. After day 357, Cr was captured by the MSWI bottom ash layer, which could be ascribed to

the immobilization of Cr(III) due to the increase of the leachate pH. As a whole, 0.93% of Cu, 0.12% of Zn, and 0.01% of Cr in the MSWI bottom ash layer were released over the study. The released heavy metals were immobilized by the sub-MSW layer. To control the heavy metal pollution, the MSWI bottom ash layer is preferentially used as the cover or intermediate cover in the landfill. The permeability coefficient must be strictly controlled if the MSWI bottom ash layer is used as the liner in the landfill, since the liner has no sub-MSW layer served as the barrier for the released heavy metals like the cover or intermediate cover.

Acknowledgement

This work was financially supported by Public Technology Applied Research Fund of Zhejiang Province Science and Technology Department with Grant No 2015C33234, Natural Science Foundation of Zhejiang province with Grant No. LQ13B070001 and Natural Science Foundation of China with Grant No. 51578356 and 41601512.

References

- Adebowale, K.O., Unuabonah, I.E., Olu-Owolabi, B.I., 2005. Adsorption of some heavy metal ions on sulfate- and phosphate-modified kaolin. *Appl. Clay Sci.* 29, 145–148.
- Benson, C.H., Barlaz, M.A., Lane, D.T., Rawe, J.M., 2007. Practice review of five bioreactor/recirculation landfills. *Waste Manage.* 27, 13–29.
- Chandler, A.J., Eighmy, T.T., Hartlen, J., Hjelmar, O., Kosson, D.S., Sawell, S.E., van der Sloot, H.A., Vehlou, J., 1997. *Municipal Solid Waste Incinerator Residues*. Elsevier Science, B.V, Amsterdam, the Netherlands.
- Chimenos, J.M., Segarra, M., Fernández, M.A., Espiell, F., 1999. Characterization of the bottom ash in municipal solid waste incinerator. *J. Hazard. Mater.* 64, 211–222.
- Dai, Y.C., Qian, G.R., Cao, Y.L., Chi, Y., Xu, Y.F., Zhou, J.Z., Liu, Q., Xu, Z.P., Qiao, S.Z., 2009. Effective removal and fixation of Cr(VI) from aqueous solution with Friedel's salt. *J. Hazard. Mater.* 170, 1086–1092.
- Inanc, B., Inoue, Y., Yamada, M., Ono, Y., Nagamori, M., 2007. Heavy metal leaching from aerobic and anaerobic landfill bioreactors of co-disposed municipal solid waste incineration bottom ash and shredded low-organic residues. *J. Hazard. Mater.* 141, 793–802.
- Lay, J.J., Li, Y.Y., Noike, T., 1998. Developments of bacterial population and methanogenic activity in a laboratory-scale landfill bioreactor. *Water Res.* 32, 3673–3679.
- Li, W.B., Yao, J., Malik, Z., Zhou, G.D., Dong, M., Shen, D.S., 2014. Impact of MSWI bottom ash codisposed with MSW on landfill stabilization with different operational modes. *BioMed Res. Int.* 2014, 1–10.
- Lo, H.M., Liao, Y.L., 2007. The metal-leaching and acid-neutralizing capacity of MSW incinerator ash co-disposed with MSW in landfill sites. *J. Hazard. Mater.* 142, 512–519.
- Lo, H.M., Lin, K.C., Liu, M.H., Pai, T.Z., Lin, C.Y., Liu, W.F., Fang, G.C., Lu, C., Chiang, C.F., Wang, S.C., Chen, P.H., Chen, J.K., Chiu, H.Y., Wu, K.C., 2009. Solubility of heavy metals added to MSW. *J. Hazard. Mater.* 161, 294–299.
- Long, Y.Y., Hu, L.F., Jiang, C.J., Fang, C.R., Wang, F.P., Shen, D.S., 2009. Releasing behavior of copper in recirculated bioreactor landfill. *Bioresour. Technol.* 100, 2419–2424.
- Long, Y.Y., Shen, D.S., Wang, H.T., Lu, W.J., 2010. Migration behavior of Cu and Zn in landfill with different operation modes. *J. Hazard. Mater.* 179, 883–890.
- Meima, J.A., van Zomeren, A., Comans, R.N.J., 1999. Complexation of Cu with dissolved organic carbon in municipal solid waste incinerator bottom ash leachates. *Environ. Sci. Technol.* 33, 1424–1429.
- Muhunthan, B., Taha, R., Said, J., 2004. Geotechnical engineering properties of incinerator ash mixes. *J. Air Waste Manage.* 54, 985–991.
- National Bureau of Statistics of China, 2015. *China Statistical Yearbook 2013*, Beijing.
- National Environmental Monitoring Centre of China, 1990. *Element Ground Value in Soils of China*. China Environmental Science Press, Beijing.
- Olsson, S., van Schaik, J.W.J., Gustafsson, J.P., Kleja, D.B., van Hees, P.A.W., 2007. Copper(II) binding to dissolved organic matter fractions in municipal solid waste incinerator bottom ash leachate. *Environ. Sci. Technol.* 41, 4286–4291.
- Paquin, P.R., Santore, R.C., Wu, K.B., Kavvas, C.D., Di Toro, D.M., 2000. The biotic ligand model: a model of the acute toxicity of metals to aquatic life. *Environ. Sci. Policy* 3, 175–182.
- Su, L.H., Guo, G.Z., Shi, X.L., Zuo, M.Y., Niu, D.J., Zhao, A.H., Zhao, Y.C., 2013. Copper leaching of MSWI bottom ash co-disposed with refuse: effect of short-term accelerated weathering. *Waste Manage.* 33, 1411–1417.
- Tessier, A., Campbell, P.G.C., Bisson, M., 1979. Sequential extraction procedure for the speciation of particulate trace metals. *Anal. Chem.* 51, 844–851.

- Travar, I., Lidelöw, S., Andreas, L., Tham, G., Lagerkvist, A., 2009. [Assessing the environmental impact of ashes used in a landfill cover construction](#). *Waste Manage.* 29, 1336–1346.
- Yao, J., Li, W.B., Kong, Q.N., Wu, Y.Y., He, R., Shen, D.S., 2010. [Content, mobility and transfer behavior of heavy metals in MSWI bottom ash in Zhejiang province, China](#). *Fuel* 89, 616–622.
- Yao, J., Kong, Q.N., Li, W.B., Zhu, H.Y., Shen, D.S., 2014. [Effect of leachate recirculation on the migration of copper and zinc in municipal solid waste and municipal solid waste incineration bottom ash co-disposed landfill](#). *J. Mater. Cycles Waste. Manag.* 16, 775–783.
- Yao, J., Kong, Q.N., Zhu, H.Y., Long, Y.Y., Shen, D.S., 2015. [Retention and leaching of nitrite by municipal solid waste incinerator bottom ash under the landfill circumstance](#). *Chemosphere* 119, 267–272.
- Youcai, Z., Lijie, S., Guojian, L., 2002. [Chemical stabilization of MSW incinerator fly ashes](#). *J. Hazard. Mater.* 95, 47–63.
- van Zomeren, A., Comans, R.N.J., 2004. [Contribution of natural organic matter to copper leaching from municipal solid waste incinerator bottom ash](#). *Environ. Sci. Technol.* 38, 3927–3932.