Journal of Zhejiang University SCIENCE A
ISSN 1673-565X (Print); ISSN 1862-1775 (Online)
www.zju.edu.cn/jzus; www.springerlink.com
E-mail: jzus@zju.edu.cn

Investigation of migration of pollutant at the base of Suzhou Qizishan landfill without a liner system*

Hai-jian XIE, Yun-min CHEN†‡, Liang-tong ZHAN, Ren-peng CHEN, Xiao-wu TANG,
Ru-hai CHEN, Han KE
(*MOE Key Laboratory of Soft Soils and Geoenvironmental Engineering, Zhejiang University, Hangzhou 310058, China*)
(*College of Civil Engineering and Architecture, Zhejiang University, Hangzhou 310058, China*)
†E-mail: chenyunmin@zju.edu.cn

Received Apr. 18, 2008; Revision accepted Aug. 19, 2008; Crosschecked Dec. 29, 2008

Abstract: We investigated migration of pollutant at the base of the Suzhou landfill after it had been operated for 13 years. The investigation was carried out by performing chemical analyses on the soil samples taken from the silty clay deposit. Concentrations of chloride, chemical oxygen demand (COD) and the heavy metals in the soil samples were determined using the standard methods. The experimental data showed that the maximum migration depth of chloride was more than 10 m, while the maximum migration depth of COD varied between 1 and 3.5 m. It is believed that the difference is attributed to the variation in diffusion rate and leachate-soil interaction. The chloride profiles also indicated that advection may be the dominant contaminant transport mechanism at this site. The total contents of Cu, Pb and Cr are very close to the background levels and the concentration values of these metals mainly are lower than the threshold values specified by the Chinese soil quality standard and the European one. The water-extractable concentrations of COD in the surface of the silty clay generally exceed the limit value specified by the Chinese standard. The concentrations of copper and chromium in pore water are 1~2 orders of magnitude less than the total concentrations of these heavy metals within the soils, implying that heavy metals are mainly adsorbed by the soil particles. Finally, remediation methods were suggested for this landfill site.

Key words: Uncontrolled landfill, Field investigation, Contaminant migration, Soil contamination, Heavy metals, Organic contaminants, Chloride
doi:10.1631/jzus.A0820299 **Document code:** A **CLC number:** TU42

INTRODUCTION

Sanitary landfills are land disposal of municipal solid waste (MSW), adopt engineering methods and have the advantage of accumulating a large volume of MSW (Hu and Chen, 2001). In China, more than 90% of municipal domestic wastes are accumulated in sanitary landfills (Zhang and Wu, 2005). Biodegradation of waste buried within landfill sites occurs as a natural process producing contaminated liquids known as leachate. The leachate typically contains high concentrations of conventional, non-conventional, and hazardous chemicals, such as biochemical oxygen demand (BOD), chemical oxygen demand (COD), as well as so-called ‘hazardous chemicals’ including heavy metals and numerous chemical compounds that may severely pollute the environment (Jensen *et al*., 1999; Christensen *et al*., 2001; Freyssinet *et al*., 2002). Therefore, the resultant pollutant plume from the leachate and its transport through the soil substrate must be examined to determine whether it poses a threat to the immediate surroundings (Yong, 2001).

The migration of the leachate constituents into the landfill basal liners or the natural soil deposits is

---

‡ Corresponding author

* Project supported by the National Natural Science Foundation of China (No. 50538080), and the National Science Fund for Distinguished Young Scholars (No. 50425825), China

usually studied by field investigation, laboratory testing and numerical modeling (Rowe *et al*., 2004; Du *et al*., 2004; 2005; Chen *et al*., 2006). Several researchers have reported the field investigation of migration of contaminants from solid waste landfills through both natural and compacted clay (Goodall and Quigley, 1977; King *et al*., 1993; Munro *et al*., 1997; Lake and Rowe, 2005). The leachate level within the studied landfills is always kept below a specific height (e.g., a height of 30 cm as regulated by USEPA (1993)). Thus, the low hydraulic conductivity ($<1\times10^{-9}$ m/s) natural clay deposits or compacted clay liners minimize the advective flux of contaminants through the clay resulting in diffusion becoming the dominant contaminant transport mechanism. However, at present, thousands of open dumps or uncontrolled landfills in Asia are generally lacking effective facilities for water management and leachate control (Karthikeyan *et al*., 2007; Swati *et al*., 2007; Sharholy *et al*., 2008). Thus, the leachate mound within the landfills as well as the leachate generation rate is quite high, particularly in humid regions (e.g., Southeast Asia) (Karthikeyan *et al*., 2007). Indeed, little field investigation has been carried out on the migration of the leachate constituents from these uncontrolled landfills into the underlying soils.

This study presented a field and laboratory investigation on the vertical and lateral migration of leachate pollutants from an unlined landfill which had been operated for 13 years. The main objective of this study is to: (1) investigate how deep the pollutants had penetrated into the soil layer over 13 years of landfill operation; (2) assess the background concentrations in the soil and determine the impact of contamination on the subsurface; and (3) approximately estimate the mobility of the heavy metals in the soil deposits. The field data presented here can be used to verify analytical models or more complex numerical models developed for solute transport through soils. The investigation results may also provide a guide for choosing effective and cost-effective remediation methods for the landfill site. Furthermore, the data reported in this study are necessary to predict further contamination trends under various conditions.

MATERIALS AND METHODS

Site investigation

The Suzhou Qizishan landfill (120°37′ E, 31°18′ N) is located about 13 km southeast of Suzhou City in China. The landfill layout is shown in Fig.1. The site is about 2 km from the nearby river (the Xu River). Landfill operations began at Qizishan in July, 1993 and it is expected to receive wastes for at least 15 years. At the time of this study, the landfill was receiving approximately 1600 t/d of MSW, which represents about 60% of the output originating from Suzhou, a city with a population of over 6.24 million. In March, 2006 Qizishan contained 4.5×10^6 t of waste that was compacted to an approximate density of 1.1 t/m^3. Fig.2 shows the main cross section of the landfill (A-A′ profile in Fig.1) as of April, 2006, when the field investigation was carried out. As shown in Fig.2, neither bottom liners nor leachate collection systems were constructed for the landfill. An injected grout curtain was installed under the retaining wall of the leachate pond to limit downstream movement of leachate (Zhan *et al*., 2008). The hydraulic conductivity of the wall was designed to be 1.0×10^{-6} cm/s. The grout curtain was made to extend to the underlying fresh rock (with a hydraulic conductivity less than 1.0×10^{-6} cm/s). The grout curtain and the fresh rock were expected to constitute a closed barrier system against the leachate in the landfill.

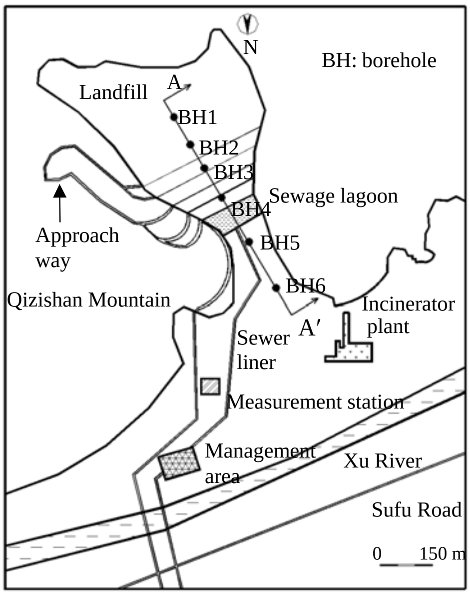

Fig.1 Location and layout of the Suzhou Qizishan landfill

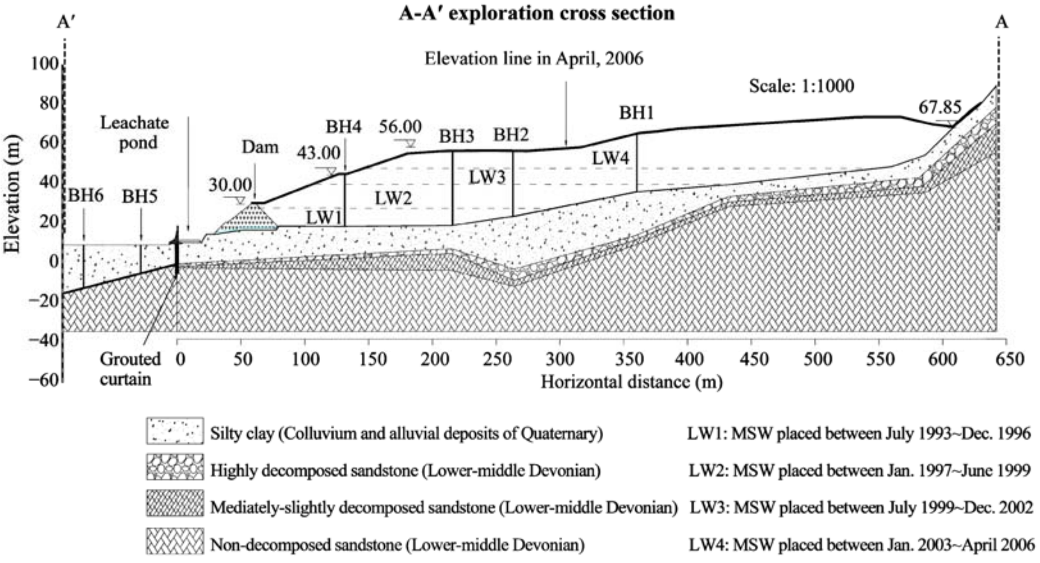

Fig.2 A-A′ cross section of the boreholes beneath the landfill site (Zhan *et al*., 2008)

The soil profile of the site typically consists of a surface layer of gravelly silty clay underlain by a gravel stratum. Below the gravel layer is a stratum of fully weathered quartzy sandstone underlain by a layer of weakly weathered quartzy sandstone and an unweathered quartzy sandstone, as shown in Fig.2. The average hydraulic conductivity of the gravel layer, fully weathered quartzy sandstone, weakly weathered quartzy sandstone, and unweathered quartzy sandstone are 1.6×10^{-3}, 3.5×10^{-4}, 1.1×10^{-4}, and 1.3×10^{-7} cm/s, respectively.

Table 1 Mineralogical and chemical data for the silty clay

Parameter	Mean
Property	
Porosity	0.41
Liquid limit (%)	35.2
Plastic limit (%)	20.8
Moisture content (%)	25
Wet unit weight (kN/m^3)	19.8
Hydraulic conductivity (cm/s)	1.1×10^{-6}
pH (soil-water ratio: 1:5 (w/v))	6.0
Mineralogy (<74 μm)	
Quartz	59.6%
Albite	15.3%
Clinochlore	5.7%
Muscovite	19.4%
Cation exchange capacity	9.1 meq/100 g

The landfill is directly founded on the silty clay layer. The thickness of this layer is estimated to vary from 0 to 20 m across the site. Ten soil samples were analyzed with respect to their mineralogical composition and their soil mechanics properties, as presented in Table 1. Atterberg limit tests indicated that the silty clay had an average liquid limit of 35.2% and an average plastic limit of 20.8%. The average moisture content of the soils was determined to be 25% and the porosity on average was 0.41. The average value of the silty clay hydraulic conductivity (measured using flexible wall hydraulic conductivity tests) was about 1.1×10^{-6} cm/s. The pH value was determined in a 1:5 (w:v) soil:de-ionized water slurry. The soils were slightly acidic with a pH value of 6.0. X-ray diffraction traces of the silty clay samples identified quartz as the primary mineral constituent with lesser proportions of albite, clinochlore and muscovite. The cation exchange capacity (CEC) was 9.1 meq/100 g.

Soil sampling

As shown in Fig.1, six boreholes (BH1 to BH6) were drilled to the silty clay deposits to take the soil samples. Four of the boreholes (BH1 to BH4) on the upstream were drilled down to the bottom of the landfill, and then entered the underlying silty clay layer to the depths of 3.9, 1.9, 1.95, and 10.35 m, respectively (Fig.2). The drilling was performed without the introduction of drilling mud and liquids. A system of steel casings was installed in each borehole to avoid the contamination of the soil samples by the leachate in the boreholes. Continuous sampling was carried out within the silty clay layer in the four boreholes (BH1 to BH4). The other two boreholes on the downstream (i.e., BH5 and BH6) were drilled to depths of 6 and 20 m, respectively. Soil sampling in these two boreholes were executed at a spacing of 2 m. It is believed that the soil samples taken from BH5 and BH6 were free from leachate contamination, at least at the deep soil layer.

All soil samples recovered from the boreholes were waxed and packed in ice at the end of each day. Following the method described by Crooks and Quigley (1984), the soil samples were stored at 10 °C in the laboratory, and chemical analyses were carried out within 3 weeks of the completion of the field work. Leachate samples were also collected from the leachate pond, and were stored at 4 °C in the laboratory. The samples were of high quality and are believed to be representative of the in situ condition.

Chemical analysis

The chemical analysis was carried out to determine the concentrations of chloride, COD, copper, lead and chromium in the soil samples. Chloride (Cl^-) was chosen to represent the inorganic contaminants in the leachate for the reason that it is not easily adsorbed by soil, difficult to decompose, and easy to measure (Kjeldsen, 1986; Chen and Wang, 1997). Another reason is that the leachate generally contains a high chloride concentration (e.g., 3638 mg/L for this site). The chemical oxygen demand (COD) represented the organic matter of the contaminants from the leachate. In addition, three common heavy metals, namely, copper, lead and chromium represented the heavy metals in the leachate. The reason for choosing these three metals is that their concentration values are relatively high in the leachate than the others (Table 2).

All reagents used in the experimental techniques described below were reagent grade for the analysis. Chinese standard solutions of Cu, Cr and Pb, and Chinese mineral standards (total digestion) were used to calibrate the quality of the obtained results by an inductively coupled plasma (ICP) atomic emission spectrometer (Varian Liberty 100, USA). Quality assurance and quality control procedures also included duplicates, blanks an sample spies. Approximately 20% of the samples were digested and spiked in duplicate to assess the precision and accuracy of the analytical procedures. Excellent agreement was obtained in the measured values of the five contaminants and the precision of the technique was in the range of 1.5%~1.9%. The statistical package used throughout this study was SPSS11.0.

Table 2 Physicochemical characteristics of leachate from the landfill

Parameter	Mean	Parameter	Mean
Cl^- (mg/L)	3638	Cd (mg/L)	0.004
SO_4^{2-} (mg/L)	15	Zn (mg/L)	0.03
K^+ (mg/L)	1666	Hg (mg/L)	<0.001
Na^+ (mg/L)	2080	COD (mg/L)	20400
Cu (mg/L)	0.16	BOD (mg/L)	8800
Pb (mg/L)	0.08	pH	8.0
Cr (mg/L)	0.29		

Determination of pore water concentrations

Pore water was extracted from the soil samples by the method described by Ohtsubo *et al.*(1995) and Du *et al.*(2005). An ion chromatograph was used to determine chloride concentrations, while an ICP atomic emission spectrometer was used for metals. COD concentration of the extract was determined using the method of dichromate described by NSPRC (1989).

Determination of total heavy metals

The total contents of Cu, Pb and Cr in the soil samples were determined following the methods described by NSPRC (1997a; 1997b). The air dried bulk soil was pulverized and a 2 g sample was successively digested at 160 °C in a 100 ml Teflon beaker with 15 ml concentrated HNO_3, 10 ml $HClO_4$, and 10 ml HF. The final residue was dissolved in 1 mol/L HCl and made up to 10 ml of solution. The solution was then centrifuged at 3000 r/min for 30 min and Cr, Cu and Pb were analyzed on the clear supernatant by a Varian Liberty 100 ICP atomic emission spectrometer.

Determination of water-extractable concentration

The water-extractable concentrations of COD were determined following the methods recommended by NSPRC (1997c). 100 g prepared air-dried soil sample together with 1 L de-ionized water were placed into a 2 L Erlenmeyer flask. Then the flask was top sealed and placed in an end-over-end rotary mixer and shaken at 29 r/min for 8 h. After 16 h of stewing, the supernatant was filtered through a 0.45-μm membrane filter. The filtrates were then analyzed for COD using the method of dichromate (NSPRC, 1989).

RESULTS AND DISCUSSION

Contaminant concentration in the leachate

The physicochemical parameters of the leachate from the landfill site are shown in Table 2. It can be seen that the concentrations of the heavy metals in the leachate are much lower than those of COD, BOD, chloride, and sodium. The concentration of COD in the leachate was 20400 mg/L, the concentrations of chloride and sodium being 8800, 3638, 2080 mg/L, respectively,’ and the concentrations of heavy metals ranging from 10^{-3} to 10^{-1} mg/L. The relatively high values of COD, BOD and chloride may be caused by the higher content of organic matter in the MSWs in China as compared to that in the developed countries (Kruempelbeck and Ehrig, 1999; Bonaparte *et al.*, 2002).

Background values of the contaminants

Fig.3 shows the vertical distributions of the five contaminants (i.e., chloride, COD, copper, lead, and chromium), which were determined from the soil samples taken from BH5 and BH6. Note that the concentrations of COD and chloride measured from the soil samples is about 1~2 orders lower than those measured from the leachate. The concentration of Cl^- in pore water for BH5 is 3~10 times greater than that for BH6. This is because BH5 is much closer to the leachate pond and the grout curtain than BH6, and the soils are more readily contaminated by the leachate.

In the upper 10 m of the soil layer, the concentration values of the contaminants show a decreasing trend with the increasing depth. However, the concentration values below a depth of 10 m basically do not vary significantly. The result indicates that the

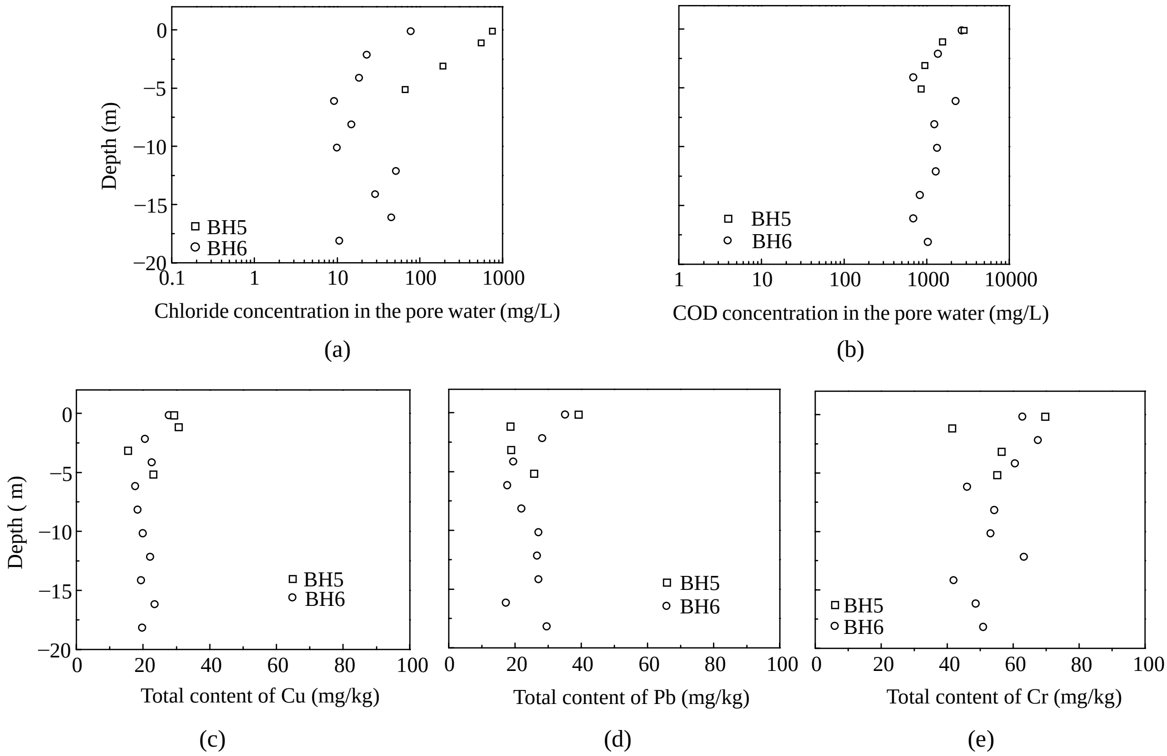

Fig.3 Vertical distribution of contaminant concentration in soil samples obtained from BH5 and BH6. (a) Chloride concentration in the pore water; (b) COD concentration in the pore water; (c) Total content of Cu; (d) Total content of Pb; (e) Total content of Cr

background values of the contaminant in the silty clay layer can be deduced from the data corresponding to depths greater than 10 m. The deduced background values in the natural clay layer are: COD, 1030 mg/L; chloride, 30 mg/L; copper, 25.3 mg/kg; lead, 29.9 mg/kg, and chromium, 51.3 mg/kg.

Vertical distribution of the five contaminants at the base of the landfill

Fig.4 shows the vertical distribution of chloride concentration, which was obtained from the four boreholes inside the landfill (i.e., BH1 to BH4). For the purpose of comparison, the background value of chloride for the site is also shown in Fig.4. It can be seen that the chloride concentration is generally greater than 1000 mg/L, which is much higher than the background value (30 mg/L). It is unexpected that the chloride concentration does not change significantly with the depth, even for a borehole extended to a depth of 10 m. The vertical distribution of chloride does appear to be an advection profile. These results seem to be inconsistent with those obtained by the other researchers (Johnson *et al.*, 1989; Kugler *et al.*, 2002). The result implies that the chloride in the leachate had transported to a depth greater than 10 m after 13 years of landfilling operations. The deep migration of chloride may be related to the high leachate mound (10~20 m) existing on the base of the landfill (Zhan *et al.*, 2008). Another reason may be that chloride is a non-reactive solute, thereby migrating faster than other contaminants such as organic contaminant and heavy metal.

Fig.5 shows the vertical distribution of COD concentration at the base of the landfill. The COD concentration generally shows a decreasing trend with the increasing depth in the four boreholes (BH1 to BH4). At the shallow depth, the COD concentration is significantly greater than the background value. However, the COD concentration decreases quickly and approaches the background value within a depth of 3 m for the three boreholes (BH2 to BH4). The COD in BH1 seemed to transport to a greater depth than that in the other three locations. The above results show that the organic contaminant tends to be retained in a relatively narrow band at the base of the landfill. As far as the data obtained from the three boreholes (BH2 to BH4) are concerned, the maximum thickness of the organic-rich band is in the order of 3 m. In other words, COD had transported into the underlying soil at a depth of up to 3 m. The finding is in agreement with the result obtained by Ahle *et al.*(1998).

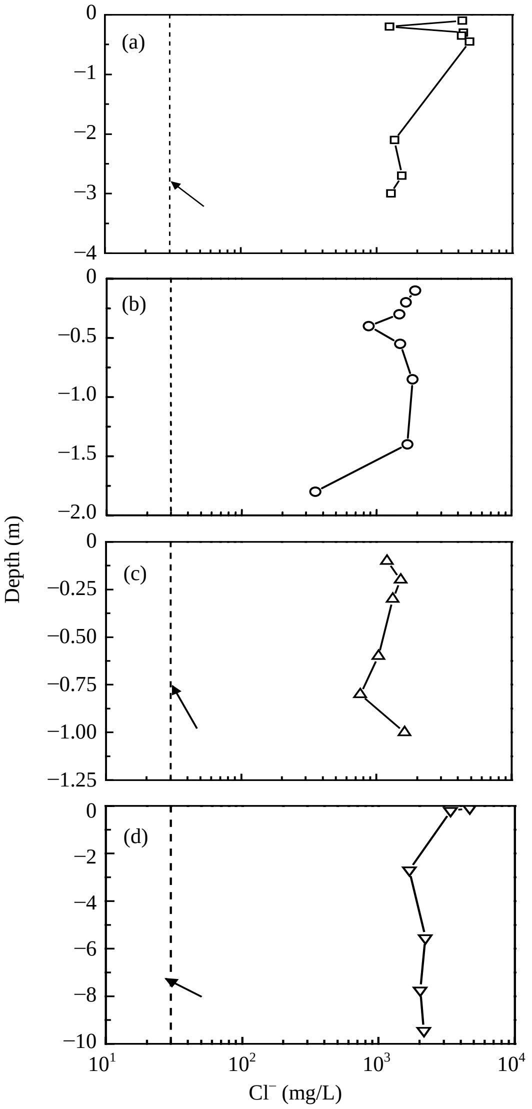

Fig.4 Vertical distribution of chloride in the silty clay beneath the landfill site after 13 years of operation. (a) BH1; (b) BH2; (c) BH3; (d) BH4. The dashed line means the background value of Cl^-=30 mg/L

It is obvious that the transport distance of COD is significantly less than that of chloride. The difference may be attributed to the following factors. First, the silty clay has a greater adsorption capacity for COD than for chloride; Second, the transport rate of COD is always much slower than that of chloride (Hrapovic and Rowe, 2002); Finally, it is noticed that the background value of COD is approximately 10%~20% of the concentrations measured at the shallow depth, while the background value of chloride is about

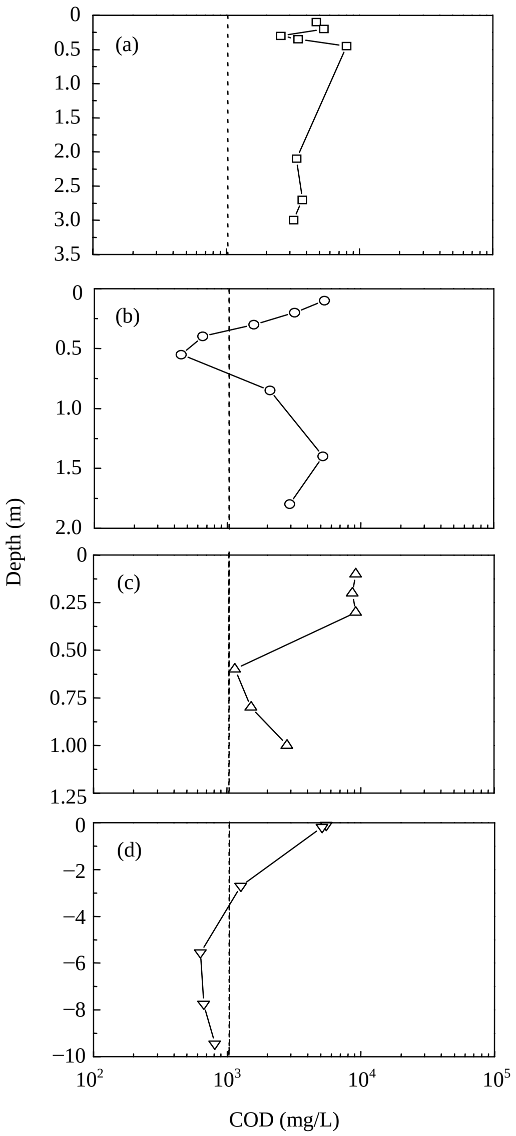

Fig.5 Vertical distribution of COD in the silty clay beneath the landfill site after 13 years of operation. (a) BH1; (b) BH2; (c) BH3; (d) BH4. The dashed line means the background value of COD=1030 mg/L

1%~5% of the concentrations measured at the shallow depth. The finding is consistent with the earlier reports by Goodall and Quigley (1977) and Munro *et al.*(1997).

Fig.6 shows vertical distribution of the total content of three heavy metals (lead, copper, and chromium), which were obtained from the four boreholes inside the landfill (i.e., BH1 to BH4). The total contents of the three heavy metals follow the order: Cr>Pb>Cu. For each of the heavy metals, the total content at the shallow depth is generally greater than the corresponding background value for the site. However, the maximal discrepancy in the two concentration values is less than 40%. For example, the maximum content of chromium for BH1 is 75 mg/kg compared to the background value of 50 mg/kg. The contamination levels for lead and chromium seem to be greater than that of copper. It also seems that lead and chromium had transported to a greater depth in comparison with copper. One possible reason for the difference is that the clay minerals in the soil samples have the greatest adsorption capacity with respect to copper.

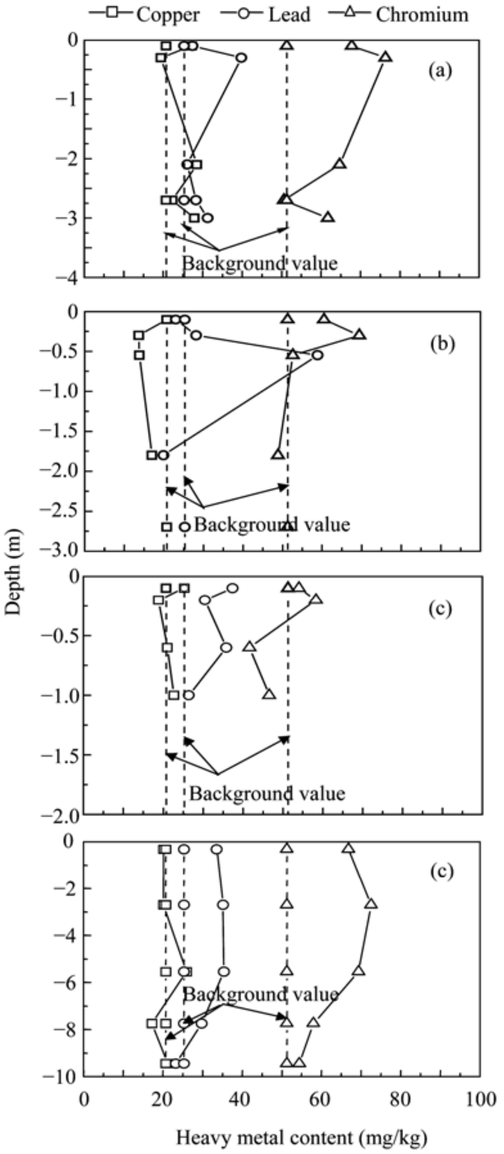

Fig.6 Vertical distribution of total heavy metal content in soil samples beneath the landfill site. (a) BH1; (b) BH2; (c) BH3; (d) BH4. The dashed lines from the left to the right mean the background values of 25.3 mg/kg, 29.9 mg/kg, and 51.3 mg/kg

Comparisons of the contaminant concentrations with the limit values

High toxicity was observed in leachate from landfills receiving MSW (Schrab *et al*., 1993). Based on multiple genotoxicity tests of leachate from MSW landfills, Schrab *et al*.(1993) found that three of four tested samples exerted genetic toxicity. The experiment results of Helma *et al*.(1996) showed that landfill leachates had higher genotoxic potency than effluents from pulp production and wastewater. Moreover, risk calculations indicated that MSW leachates may present as great a cancer risk as those from co-disposal and industrial solid waste landfills (Schrab *et al*., 1993). Therefore, high attention should be directed to the toxic leachate constituents, which may migrate into soil and groundwater and pose a health risk to humans via food chain contamination (Bruder-Hubscher *et al*., 2002).

In this subsection, the contaminant concentrations measured from the top 3 m soil layer beneath the landfill were compared with the soil quality standards to assess the extent of soil contamination at this site. The allowable values of the contaminants for the Chinese and European standards are also stated in Table 3. The allowable values of heavy metals are obtained from soil environmental quality of China (NSPRC, 1995) and the European Union soil quality standards (Marzougui and Mammou, 2006), while the allowable value of COD is obtained from the Profession Standard of Environmental Protection of China (PSEPPRC, 2005).

It can be seen that the concentrations of chromium and copper do not exceed the allowable values regulated in Chinese standards, even at the shallow depth. The concentration of lead exceeds the first-level standard value of NSPRC (1995) only at the shallow depth. It can also be seen that the concentrations for all the three heavy metals are less than the allowable values regulated by the European standard (Marzougui and Mammou, 2006). The above comparison indicates that unallowable soil contamination with respect to heavy metals occurred only at the shallow depth beneath the bottom of the landfill. The result is consistent with those reported by Yanful *et al*.(1988) and Marzougui and Mammou (2006). The above finding is understandable because the concentrations of the heavy metals in the leachate were much lower than those of chloride and COD.

Table 3 Pb, Cu, Cr, and COD concentrations in the soils with depths of 0~3 m compared to the specified threshold values

	Depth (m)	Cu (mg/kg)	Pb (mg/kg)	Cr (mg/kg)	COD (mg/L)
BH1	0~0.2	27.2	27.5	67.8	347
	0.2~0.4	19.4	39.8	76.3	176
	2.0~2.2	28.6	26.1	64.7	69
BH2	0~0.2	20.8	23.0	60.5	125
	0.45~0.65	13.9	58.9	52.6	16
	1.7~1.9	17.0	19.9	48.8	74
BH3	0~0.2	25.3	37.4	54.0	152.5
	0.5~0.7	21.1	35.8	41.7	28
	0.9~1.1	22.6	26.4	46.6	46
BH4	0~0.2	20.1	33.6	66.3	134.5
	2.6~2.8	20.1	35.2	72.6	16
BH5	0~0.2	29.1	39.0	69.4	62
	1.0~1.2	30.5	18.4	41.2	34
BH6	0	27.6	35.0	62.5	58
	2	20.4	28.0	67.2	30
A		35	35	90	−
B		30~60	50~100	50~100	−
C		−	−	−	60

A: Chinese soil quality standard (first-level values); B: European soil quality standard; C: Chinese professional standard

The water-extractable concentration of COD measured from the top 0.2 m soil layer generally exceeds the allowable value, 60 mg/L, specified in HJ85-2005 (PSEPPRC, 2005). The concentration of COD for BH1 is significantly greater than the allowable value down to a depth of 0.4 m, and keeps above the allowable level down to a depth of 2.2 m. It is also noted that the maximum concentration of COD in the two boreholes downstream of the grout curtain (i.e., BH5 and BH6) is close to the allowable value. The result indicates that the surfacial soil downstream of the grout curtain had been contaminated by the leachate. The surface contamination may be caused by occasional overflow of leachate from the leachate pond. Further studies should be performed to assess the extent of soil contamination caused by the leachate.

Comparisons between pore water concentration and total content for the heavy metals

In addition, to analyze the mobility of heavy metals retained within the sample, determinations of heavy metal in the pore water are required. This type of information is useful not only in determining the kinds of soil materials to be used in soil-engineered barriers, but also in assessing the environmental mobility of these kinds of inorganic pollutants (Yong, 2001). We can also develop strategies for remediation, based on the knowledge of the distribution of the heavy metal pollutants (Mulligan *et al*., 2001).

The metals concentration profiles in the pore water together with the total content of the metals are shown in Fig.7. The data about Pb concentrations in the pore water were not given in the figure for the reason that the value is significantly lower (<0.05 mg/kg). Results indicate that both Cr and Cu are minimal quantities in the pore water, with Pb being almost zero throughout the length of the sample. It can also be seen that the copper and chromium concentrations in pore water are 1~2 orders of magnitude less than the total concentrations of these heavy metals within the soils. For example, as shown in Fig.7a, the chromium concentration in pore water is approximately 2 orders of magnitude less than the total content of this metal in the soils. Similarly, the copper concentration in pore water is approximately 1.6 orders of magnitude less than the total copper in the soils (Fig.7b). It is argued that the heavy metals are mainly adsorbed by the soil particles and that the heavy metals in the pore water can be neglected.

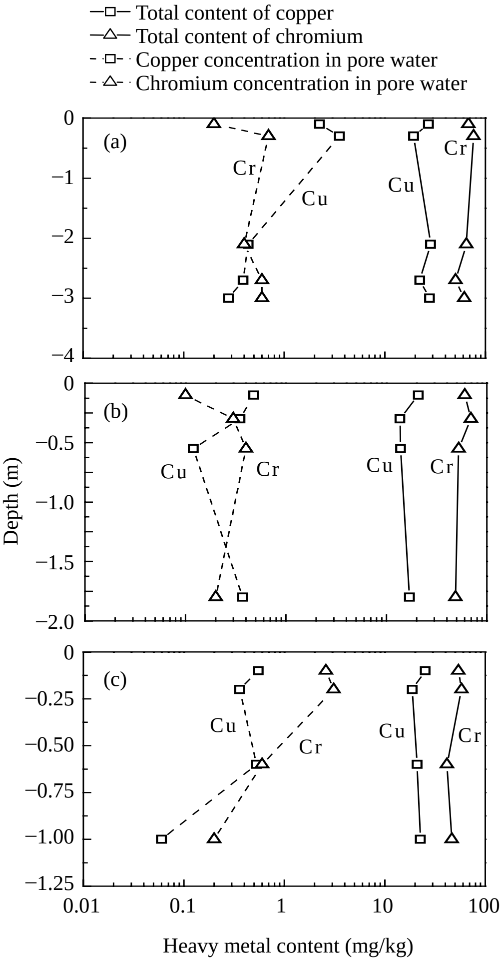

Fig.7 Comparisons between pore water and total content of the heavy metals in soil samples beneath the waste. (a) BH1; (b) BH2; (c) BH3

CONCLUSION

Contaminant profiles below the Suzhou Qizishan landfill site have been established by measuring the concentrations of COD, chloride and heavy metals in the soil samples obtained from the Shelby tubes. Specific conclusions drawn from the site are as follows:

(1) The maximum depth of migration of chloride was beyond the depths at each of the boreholes, while the migration distance of COD varied between 1~3.5 m for the four boreholes (BH1~BH4) due to be caused by variations in diffusion rates and leachate-soil interaction such as adsorption. The chloride distribution profiles also indicate that advective transport may be the dominant mechanism controlling solute transport through the underlying soils. Though the maximum migration distances of lead and chromium reached 10 and 8 m, respectively, the concentration values of the total Cu, Pb and Cr are very close to the natural geological background levels.

(2) The total contents of the heavy metals are mainly lower than the limit values given by the Chinese soil quality standard and the European one, suggesting that soils below the landfill had been slightly contaminated by the heavy metals. The water-extractable concentration values of COD in the surface of the soils are generally larger than the limit values, indicating that the surfacial soil downstream of the site had been contaminated by the pollutants. Results also show that the copper and chromium concentrations in pore water are 1~2 orders of magnitudes less than the total concentrations of these heavy metals within the soils, indicating that the heavy metals are mainly adsorbed by the soil particles

and that the heavy metals in the pore water can be neglected.

(3) Retro-fitted measures should be taken to lower the high leachate level and control the leachate generation rate. For example, the installing of pumping vertical walls is suggested to lower the leachate level within the landfill and an impermeable clay layer should be provided for the landfill cover system to decrease the amount of the leachate generated in the landfill. Finally, it is recommended that more effective vertical barrier walls (lower hydraulic conductivity and larger thickness) should be constructed for further preventing the leachate escape.

References

Ahel, M., Mikac, N., Cosovic, B., Prohic, E., Soukup, V., 1998. The impact of contamination from a municipal solid waste landfill (Zagreb, Croatia) on underlying soil. *Water Science and Technology*, **37**(8):203-210. [doi:10.1016/S0273-1223(98)00260-1]

Bonaparte, R., Daniel, D., Koerner, R.M., 2002. Assessment and Recommendations for Improving the Performance of Waste Containment Systems. EPA Report, Co-operative Agreement Number CR-821448-01-0.

Bruder-Hubscher, V., Lagarde, F., Leroy, M.J.F., Coughanowr, C., Enguehard, F., 2002. Application of a sequential extraction procedure to study the release of elements from municipal solid waste incineration bottom ash. *Analytica Chimica Acta*, **451**(2):285-295. [doi:10.1016/S0003-2670(01)01403-9]

Chen, P.H., Wang, C.Y., 1997. Investigation into municipal waste leachate in the unsaturated zone of the red soil. *Environment International*, **23**(2):237-245. [doi:10.1016/S0160-4120(97)00010-X]

Chen, Y.M., Xie, H.J., Ke, H., Tang, X.W., 2006. Analytical solution of contaminant diffusion through multilayered media. *Chinese Journal of Geotechnical Engineering*, **28**:521-524 (in Chinese).

Christensen, T.H., Kjeidsen, P.L., Bjerg, D.L., Jensen, D.L., Christensen, J.B., Baun, A., Albrechtsen, H., Heron, G., 2001. Biochemistry of landfill leachate plumes. *Applied Geochemistry*, **16**(7-8):659-718. [doi:10.1016/S0883-2927(00)00082-2]

Crooks, V.E., Quigley, R.M., 1984. Saline leachate migration through clay: a comparative laboratory and field investigation. *Canadian Geotechnical Journal*, **21**:349-362.

Du, Y.J., Hayashi, S., Xu, Y.F., 2004. Some factors controlling the adsorption of potassium ions on clayey soils. *Applied Clay Science*, **27**(3-4):209-213. [doi:10.1016/j.clay.2004.06.003]

Du, Y.J., Hayashi, S., Liu, S.Y., 2005. Experimental study of migration of potassium ion through a two-layer soil system. *Environmental Geology*, **48**(8):1096-1106. [doi:10.1007/s00254-005-0050-y]

Freyssinet, P., Piantone, M., Azaroual, Y.B., Clozel-Leloup, Y.I., Guyonnet, D., Baubron, J.C., 2002. Chemical changes and leachate mass balance of municipal solid waste bottom ash submitted to weathering. *Waste Management*, **22**(2):159-172. [doi:10.1016/S0956-053X(01)00065-4]

Goodall, D.C., Quigley, R.M., 1977. Pollutant migration from sanitary landfill sites near Sarnia, Ontario. *Canadian Geotechnical Journal*, **14**:223-236.

Helma, C., Mersch-Sundermann, V., Houk, V.S., Glasbrenner, U., Klein, C., Wenquing, L., Kassie, F., Schulte-Hermann, R., Knasmiller, S., 1996. Compartive evaluation of four bacterial assays for the detection of genotoxic effects in the dissolved water phases of aqueous matrices. *Environmental Science and Technology*, **30**(3):897-907. [doi:10.1021/es950355w]

Hrapovic, L., Rowe, R.K., 2002. Intrinsic degradation of volatile fatty acids in laboratory compacted clayey soil. *Journal of Contaminant Hydrology*, **58**(3-4):221-242. [doi:10.1016/S0169-7722(02)00038-4]

Hu, M.Y., Chen, Y.M., 2001. Engineering aspects of landfilling municipal solid waste. *Journal of Zhejiang University SCIENCE*, **2**:34-40.

Jensen, D.L., Ledin, A., Christensen, T.H., 1999. Speciation of heavy metals in landfill-leachate polluted groundwater. *Water Research*, **33**(11):2642-2650. [doi:10.1016/S0043-1354(98)00486-2]

Johnson, R.L., Cherry, J.A., Pankow, J.F., 1989. Diffusive contaminant transport in natural clay: A field example and implications for clay-lined waste disposal sites. *Environmental Science and Technology*, **23**(3):340-349. [doi:10.1021/es00180a012]

Karthikeyan, O.P., Swati, M., Nagendran, R., Joseph, K., 2007. Performance of bioreactor landfill with waste mined from a dumpsite. *Environmental Monitoring and Assessment*, **135**(1-3):141-151. [doi:10.1007/s10661-007-9709-z]

King, K.S., Quigely, R.M., Fernandez, F., Reades, D.W., Bacopoulos, A., 1993. Hydraulic conductivity and diffusion monitoring of the Keele Valley Landfill liners, Maple, Ontario. *Canadian Geotechnical Journal*, **30**:124-134.

Kjeldsen, P., 1986. Attenuation of Landfill Leachate in Soil and Aquifer Material. PhD Thesis, Technical University of Denmark.

Kruempelbeck, I., Ehrig, H.J., 1999. Long Term Behavior of Municipal Solid Waste Landfills in Germany. Proceedings of the 7th International Landfill Symposium, S. Margherita di Pula, Gagliari, Italy, p.27-36.

Kugler, H., Ottner, F., Froeschl, H., Adamcova, R., Schwaighofer, B., 2002. Retention of inorganic pollutants in clayey base sealings of municipal landfills. *Applied Clay Science*, **21**(1-2):45-48. [doi:10.1016/S0169-1317(01)00091-6]

Lake, C.B., Rowe, R.K., 2005. The 14-year performance of a compacted clay liner used as a part of a composite liner system for a leachate lagoon. *Geotechnical and Geological Engineering*, **23**(6):657-678. [doi:10.1007/s10706-004-8815-8]

Marzougui, A, Mammou, A.B., 2006. Impact of the dumping site on the environment: Case of the Henchir EI Yahoudia Site, Tunis, Tunisia. *Comptes Rendus Geosciences*, **338**(16):1176-1183. [doi:10.1016/j.crte.2006.09.020]

Mulligan, C.N., Yong, R.N., Gibbs, B.F., 2001. The Use of Selective Extraction Procedures for Soil Remediation. Proceedings of International Symposium on Suction, Swelling, Permeability and Structure of Clays, Balkema, Rotterdam, p.377-384.

Munro, I.R.P., MacQuarrie, K.T.B., Valsangkar, A.J., Kan, K.T., 1997. Migration of landfill leachate into a shallow clayey till in southern New Brunswick: A field and modeling investigation. *Canadian Geotechnical Journal*, **34**(2):204-219. [doi:10.1139/cgj-34-2-204]

NSPRC (National Specifications of the People's Republic of China), 1989. Water Quality—Determination of the Chemical Oxygen Demand—Dichromate Method. GB11914-89 (in Chinese).

NSPRC (National Specifications of the People's Republic of China), 1995. Environmental Quality Specifications for Soils. GB 15618-1995 (in Chinese).

NSPRC (National specifications of the People's Republic of China), 1997a. Soil Quality—Determination of Copper, Zinc─Flame Atomic Absorption Spectrophotometry. GB/T 17138-1997 (in Chinese).

NSPRC (National Specifications of the People's Republic of China), 1997b. Soil Quality—Determination of Lead, Cadmium─Graphite Furnace Atomic Absorption Spetrophotometry. GB/T 17141-1997 (in Chinese).

NSPRC (National Specifications of the People's Republic of China), 1997c. Test Method Specification for Leaching Toxicity of Solid Wastes—Horizontal Vibration Extraction Procedure. GB 5086.2-1997 (in Chinese).

Ohtsubo, M., Egashira, K., Kashima, K., 1995. Depositional and post-depositional geochemistry, and its correction with the geotechnical properties of marine clays in Ariake Bay, Japan. *Geotechnique*, **45**:509-523.

PSEPPRC (Profession Standard of Environmental Protection of People's Republic of China), 2005. Technical Specification of Solid Waste Cleaning for Reservoir Bed of the Three Gorges on the Yangtze River. HJ 85-2005 (in Chinese).

Rowe, R.K., Quigley, R.M., Brachman, R.W.I., Booker, J.R., 2004. Barrier Systems for Waste Disposal Facilities. E&FN Spon, London.

Schrab, G.E., Brown, K.W., Donnelly, K.C., 1993. Acute and genetic toxicity of municipal landfill leachate. *Water Air and Soil Pollution*, **69**:99-112.

Sharholy, M., Ahmad, K., Mahmood, G., Trivedi, R.C., 2008. Municipal solid waste management in Indian cities—A review. *Waste Management*, **28**(2):457-467. [doi:10.1016/j.wasman.2007.02.008]

Swati, M., Karthikeyan, O.P., Joseph, K., Nagendran, R., 2007. Landfill bioreactor: A biotechnological solution for waste management. *Journal of Science & Industrial Research*, **66**:670-674.

USEPA (United States Environmental Protection Agency), 1993. Solid Waste Disposal Facility Criteria—Technical Manual, EPA530-R-93-017.

Yanful, E.K., Quigley, R.M., Nesbitt, H.W., 1988. Heavy metal migration at a landfill site, Sarnia, Ontario, Canada—2: metal partitioning and geotechnical implications. *Applied Geochemistry*, **3**(6):623-629. [doi:10.1016/0883-2927(88)90094-7]

Yong, R.N., 2001. Geoenvironmental Engineering: Contaminated Soil, Pollutant Fate, and Mitigation. CRC Press, New York.

Zhan, L.T., Chen, Y.M., Lin, W.A., 2008. Shear strength characterization of municipal solid waste at the Suzhou landfill, China. *Engineering Geology*, **97**(3-4):97-111. [doi:10.1016/j.enggeo.2007.11.006]

Zhang, P., Wu, Z.C., 2005. Municipal sludge as landfill barrier material. *Journal of Environmental Sciences*, **17**:474-477.