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# Trace metal contamination of sediments in an e-waste processing village in China

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## Abstract

This study examined trace metal contamination of sediments in Guiyu, China where primitive e-waste processing activities have been carried out. It was found that some river sediments in Guiyu were contaminated with Cd (n.d.–10.3 mg/kg), Cu (17.0–4540 mg/kg), Ni (12.4–543 mg/kg), Pb (28.6–590 mg/kg), and Zn (51.3–324 mg/kg). The <sup>206</sup>Pb/<sup>207</sup>Pb and <sup>208</sup>Pb/<sup>207</sup>Pb ratios of the Pb-contaminated sediments of Lianjiang ( $1.1787 \pm 0.0057$  and  $2.4531 \pm 0.0095$ , respectively) were lower than those of Nanyang River ( $1.1996 \pm 0.0059$  and  $2.4855 \pm 0.0082$ , respectively), indicating a significant input of non-indigenous Pb with low <sup>206</sup>Pb/<sup>207</sup>Pb and <sup>208</sup>Pb/<sup>207</sup>Pb ratios. Copper, Pb and Zn in the non-residual fractions noticeably increased in the contaminated sediments compared to those in the uncontaminated sediments. A genuine concern is associated with potential transport of the contaminated sediments downstream and enhanced solubility and mobility of trace metals in the non-residual fractions.

Keywords: Trace metals; Pb isotopes; Sequential chemical speciation; Sediment; E-waste; China

## 1. Introduction

In this paper, we would like to emphasize on trace metal contamination arisen from primitive recycling and processing of electronic and computer wastes, also known as e-waste. E-waste typically consists of a broad range of electrical and electronic products, including computers, mobile phones, televisions, and their components, such as printed circuit boards. Disposal of e-waste is an emerging global environmental issue, as these wastes have become one of the fastest growing waste

types in some parts of the world, including the U.S. and Europe (National Safety Council, 1999, and Schmidt, 2002). Complexity in the design and composition of these electrical and electronic devices renders them undesirable for recycling and reuse for technical and economic reasons (Berkhout and Hertin, 2004, Realff et al., 2004 and Toffel and Horvath, 2004]).

These devices often contain toxic trace metals, such as Hg, Pb and Cu (Jang and Townsend, 2003). If not disposed of appropriately, they can become a source of trace metal contaminants. This is particularly worrying in some developing countries, including China. It has been reported that they have received a large volume of e-waste from developed countries for recycling and disposal (The Basal Action Network, 2002 and USA Today, 2002). Even though the intention of salvaging reusable parts from these waste products is admirable, the processes and techniques used during the recycling activities are often primitive and lack pollution control measures. In developing countries where environmental enforcement may not be sufficiently stringent, excessive levels of trace metals can be emitted. Personal visits to Guiyu, one of the e-waste processing villages in China, confirmed that the primitive recycling processes/techniques included open burning of plastics (to reduce waste volume) and copper wires (to salvage valuable metals, e.g. copper), strong acid leaching of printed circuit boards (to recover precious metals) from which the waste acids were discharged into nearby streams, and grilling of printed circuit boards over honeycomb-coal fires to melt solder (to allow collection of electronic components, e.g. diodes and resistors). It was observed that all of these operations were conducted manually with no or very little personal protective equipment or pollution control measures. Despite the potentially serious and long-term environmental and health implications, scientific information on trace metal contamination associated with these primitive e-waste recycling activities in Guiyu and elsewhere remains limited.

To understand the extent and nature of environmental degradation caused by these e-waste processing activities, this paper aimed to study trace metal contamination of river sediments in Guiyu. The objectives of the study were (1) to establish a basic understanding of the level and extent of trace metal contamination in sediments in Guiyu, (2) to study the spatial distribution and dispersion of trace metals in the aquatic environment, and (3) to examine mobility, solubility and potential bioavailability of trace metals in the environment. It is hoped that the findings will not only provide valuable information on the environmental quality of Guiyu, but also present a scientific perspective of the environmental effects of primitive e-waste recycling so that further regulatory and scientific attention can be drawn to the issue.

## 2. Materials and methods

### 2.1 Study area

The study area of Guiyu is one of the known e-waste processing villages in China. Located in Southern China, Guiyu is in the Chaozhou region of the Guangdong Province, located at a latitude and longitude of 23.3 N and 116.3 E (Fig. 1). Since it is situated in a subtropical region, its climate is characterized as mild and warm with plentiful rainfall in the summer months (April–September) and dry and cool in the winter months (October and March). Recycling of e-waste in Guiyu has taken place for nearly 10 years, with more than million tonnes of computer waste handled each year (Lee, 2002). Amid these activities, active aqua- and agricultural operations can still be found in rivers and farmlands surrounding the area.

### 2.2 Sample collection

To study the extent of trace metal contamination of the river system, grab sediment samples (0–15 cm) were collected from two rivers in Guiyu in August and September 2005. These two rivers were Lianjiang and Nanyang River, hereafter referred to as LJ and NR, respectively. The depth of the two rivers generally varied from 3 to 6 m. A total of 15 and 10 sediment samples were collected from LJ and NR, respectively, using a stainless steel collector. The sediments were immediately wrapped in aluminium foil, double-bagged in Ziploc<sup>®</sup> plastic bags and stored in an iced cooler. Upon arrival in the laboratory, the sediments were immediately refrigerated at 4 °C until pre-treatment.

### 2.3. Laboratory analyses

The moist sediments were first homogenized in a plastic container and freeze-dried.

The dried samples were further homogenized, and a portion of ~50 g from each of the samples was ground to fine particle size manually in an agate mortar with a pestle. The remains of the dried sediment samples were reserved for analyses of pH and particle size distribution. The dried and powdered samples were used for total organic carbon (TOC) analysis and chemical characterization, including total elemental concentrations, Pb isotopic composition, and sequential chemical distribution of trace metals, as described below. The pH of the sediments was measured by a calibrated pH

meter (Schott handylab, pH 11), as described previously by Allen et al. (1974) and Wong et al. (2002). Briefly, 4.0 g of the sediments (<2 mm) was placed in centrifuge tubes with 10.0 ml deionized water. The mixtures were vortexed and shaken for 30 min. pH of the solutions was measured after centrifugation. The particle size distribution of the sediments was determined by Laser Particle Size Analyser (Coulter LS-100; Beckman, Fullerton, CA), and the particle size fractions were defined according to the British Soil Classification System. TOC of the sediments was analyzed using an automated TOC analyzer (Shimadzu TOC-5000 with SSM 5000A).

For total elemental concentrations, the dried and powdered sediments were dissolved using a combination of concentrated nitric acid (HNO<sub>3</sub>) and perchloric acid (HClO<sub>4</sub>). It should be noted that elements that are strongly associated with or entrapped within silicate may not be fully recovered as this “pseudo” total acid digestion did not involve the use of hydrofluoric acid (HF). Trace metal concentrations (Cd, Co, Cu, Ni, Pb, V and Zn) of the digested solutions were determined using Inductively Coupled Plasma-Atomic Emission Spectrometry (ICP-AES, Perkin Elmer Optima 3300DV). The analytical procedure, quality control and assurance measures were implemented according to the methods described previously in Wong et al. (2002) and Wong and Li (2004). In addition to blanks and replicates, a certified reference material (CRM PACS-2 Marine Sediment Reference Material for Trace Metals and Other constituents) from National Research Council Canada was inserted in the analytical procedure to evaluate the effectiveness of the digestion procedure. The percent recoveries of Cd, Co, Cu, Ni, Pb, V and Zn from CRM PACS-2 were 84.8, 94.2, 100.9, 74.1, 83.9, 75.6 and 85.5%, respectively.

For Pb isotopic composition of sediments, the digested solutions were diluted using high-purity 5% (v/v) HNO<sub>3</sub> and analyzed using Inductively Coupled Plasma-Mass Spectrometry (ICP-MS, Perkin Elmer Elan 6100 DRC<sup>Plus</sup>). Instrumental parameters of 250 sweeps/reading, 10 readings and a dwell time of 50 ms were set. The relative standard deviation (RSD) of the 10 measurements of each of the solutions was generally <0.4%. An international standard reference material (NIST 981 Common Lead Isotopic Standard from the National Institute for Standards and Testing) was used for calibration and quality control. The measured average ratios of <sup>206</sup>Pb/<sup>207</sup>Pb and <sup>208</sup>Pb/<sup>207</sup>Pb ( $1.0937 \pm 0.0010$  and  $2.3694 \pm 0.0049$ , respectively) of the standard reference material were in close agreement with the certified standard values (1.0933 and 2.3704, respectively).

In order to evaluate mobility and potential bioavailability of Cu, Pb and Zn, chemical speciation of these trace metals in selected sediment samples was analyzed using a

modified Tessier sequential chemical extraction ( Tessier et al., 1979 and Li et al., 1995). The analytical and quality control procedure used in this study followed that previously described in Wong et al. (2002) and Wong and Li (2004). Briefly, Tessier's sequential chemical extraction is an operationally-defined analytical procedure that differentiates trace metals by solubility, reactivity and general affiliation with a specific chemical group using various chemical reagents. The extraction procedure used in this study defined trace metals into five sequential chemical fractions: (1) readily soluble and exchangeable (exchangeable) (2) carbonate-bound, specifically adsorbed, and weak organic and inorganic complexes (carbonate) (3) bound to iron and manganese oxides (Fe–Mn oxide), (4) bound to stable complexes of organic and/or sulphide (organic), and (5) residual fraction containing primary and secondary minerals held within their crystal structure (residual). It is generally acknowledged that the reactivity and potential bioavailability of trace metals increase with increasing solubility (Li et al., 1995). The Cu, Pb and Zn concentrations of the extracted solutions were measured by ICP-AES (Perkin Elmer Optima 3300DV). The metal concentrations of procedural blanks were generally <3% of the absolute concentrations of the analytical samples. The recovery rates calculated from the sum of the five fractions over the total trace metal concentrations determined by the “pseudo” total acid digestion ranged between 85 and 110%.

### 3. Results

#### 3.1. Physical characteristics of the sediments

Table 1 summarizes the physical characteristics of the sediments collected from LJ and NR in Guiyu. At the time of sample collection, the sediments from both of the rivers appeared to be very dark in colour and muddy or clay-like. The sediments also released offensive H<sub>2</sub>S-like odour, suggesting that they were anoxic (oxygen-depleted). From the analytical measurements, the sediments collected from LJ and NR were generally slightly acidic (mean pH of  $6.44 \pm 0.41$  and  $6.40 \pm 1.19$ , respectively) with pH ranging from 6.03 to 7.75 and 4.63 to 7.74, respectively. The sediments from LJ ( $37.2 \pm 20.3$  mg/kg) were considerably richer in TOC than those from NR ( $11.3 \pm 3.78$  mg/kg). The particle size distribution of the LJ sediments showed that the sediments consisted of  $55.2 \pm 9.15\%$  of silt (>2–63  $\mu\text{m}$ ),  $21.0 \pm 4.81\%$  of fine sand (>63–200  $\mu\text{m}$ ),  $17.8 \pm 10.0\%$  of medium sand (>200–800  $\mu\text{m}$ ), and  $5.98 \pm 1.10\%$  of clay (<2  $\mu\text{m}$ ). Whereas, the NR sediments were

predominantly made of silt particles ( $78.8 \pm 8.74\%$ ) with minor fractions of clay ( $12.1 \pm 6.60\%$ ), fine and medium sand ( $8.37 \pm 4.05$  and  $0.75 \pm 0.57\%$ , respectively).

### 3.2. Chemical characteristics of the sediments

Means and ranges of trace metal concentrations of the sediments collected from LJ and NR are presented in Table 2. Regulatory guidelines from Canada and the Netherlands are also included in Table 2 since China does not have its own regulatory guideline for sediments. Mean ( $\pm$ standard deviation) concentrations of Cd, Co, Cu, Ni, Pb, V and Zn concentrations in NJ sediments were  $4.09 \pm 3.92$ ,  $15.5 \pm 5.40$ ,  $1070 \pm 1210$ ,  $181 \pm 156$ ,  $230 \pm 169$ ,  $54.3 \pm 5.92$  and  $324 \pm 143$  mg/kg, respectively. Meanwhile, Co, Cu, Ni, Pb, V and Zn concentrations of NR sediments were measured at  $12.7 \pm 3.63$ ,  $65.1 \pm 101$ ,  $25.2 \pm 10.5$ ,  $47.3 \pm 13.8$ ,  $62.6 \pm 15.1$  and  $107 \pm 54.9$  mg/kg, respectively, and Cd concentrations of NR sediments ranged between undetected and  $0.57$  mg/kg. The relatively large standard deviations of trace metal concentrations of LJ sediments suggested that the LJ sediments were heterogeneous in terms of trace metal distribution. In general, it was observed that the LJ sediments were significantly enriched with trace metals compared to NR, with differences in Cd, Cu, Ni, Pb and Zn concentrations between the two rivers being the most noticeable. Cu, Ni, Pb and Zn concentrations of LJ sediments were approximately 16, 7, 5 and 3-fold those of NR sediments.

When compared with Probable Effect Levels (PELs) of the Canadian Environmental Quality Guidelines recommended by Canadian Council of Ministers of the Environment (CCME, 1997), mean Cd, Cu, Pb and Zn concentrations of the LJ sediments would be considered contaminated with a potential to cause toxic ecological effects. In comparison, trace metal concentrations of the NR sediments were generally below the corresponding PELs. The intervention values recommended by the Netherlands Sediment Quality Guidelines (Department of Soil Protection, Netherlands, 1994) represent the maximum threshold values of trace metals in sediments, above which remedial actions should be considered. A comparison with the intervention values showed that twelve out of the fifteen LJ sediments and one of the ten NR sediments exceeded the intervention Cu concentration ( $190$  mg Cu/kg; see Table 2). Moreover, Pb concentration of one NR sediment (namely NR-10;  $590$  mg Pb/kg) exceeded the corresponding intervention value ( $530$  mg Pb/kg). In addition to Cu and Pb, Ni concentrations of seven LJ sediments were also elevated compared to

the intervention Ni concentration of 210 mg/kg. Hence, the results strongly indicate that further investigation is needed, especially for LJ. Fig. 2 depicts the distribution of Cu, Pb and Zn in the sediments collected from LJ and NR. Fig. 2a shows that Cu concentrations in LJ were significantly higher and more heterogeneous than those in NR. Similarly, Pb and Zn concentrations in the LJ sediments were also considerably more elevated than those in the NR sediments. The distribution of Cu, Pb and Zn in the LJ sediments exhibits similar fluctuations along the river, possibly indicating points of anthropogenic discharge and/or locations of preferential trace metal deposition in the sediments.

### 3.3. Pb isotopic composition

Fig. 3 depicts the distribution of Pb concentrations,  $^{206}\text{Pb}/^{207}\text{Pb}$  and  $^{208}\text{Pb}/^{207}\text{Pb}$  ratios of the sediments from LJ and NR. The  $^{206}\text{Pb}/^{207}\text{Pb}$  and  $^{208}\text{Pb}/^{207}\text{Pb}$  ratios of LJ sediments ranged from 1.1693 to 1.1896 (mean:  $1.1787 \pm 0.0057$ ) and 2.4358 to 2.4678 (mean:  $2.4531 \pm 0.0095$ ), respectively. The  $^{206}\text{Pb}/^{207}\text{Pb}$  and  $^{208}\text{Pb}/^{207}\text{Pb}$  ratios of NR sediments ranged from 1.1701 to 1.2099 and 2.4516 to 2.5008, respectively. Excluding one NR sediment (namely NR-7), the mean  $^{206}\text{Pb}/^{207}\text{Pb}$  and  $^{208}\text{Pb}/^{207}\text{Pb}$  ratios of the NR sediments ( $N = 9$ ) were  $1.1996 \pm 0.0059$  and  $2.4855 \pm 0.0082$ , respectively. The distribution of Pb concentrations and  $^{206}\text{Pb}/^{207}\text{Pb}$  ratios shown in Fig. 3a indicated that Pb concentrations of the sediments increased with decreasing  $^{206}\text{Pb}/^{207}\text{Pb}$  ratios. When the  $^{206}\text{Pb}/^{207}\text{Pb}$  and  $^{208}\text{Pb}/^{207}\text{Pb}$  ratios of sediments from the two rivers were plotted, the Pb isotopic ratios for the two rivers were noticeably different from each other (see Fig. 3b). It was obvious that the  $^{206}\text{Pb}/^{207}\text{Pb}$  and  $^{208}\text{Pb}/^{207}\text{Pb}$  ratios of the NR sediments were generally higher than those of LJ sediments and the sediments from the two rivers formed two distinct groups. The linear relationship between the  $^{206}\text{Pb}/^{207}\text{Pb}$  and  $^{208}\text{Pb}/^{207}\text{Pb}$  ratios of the sediments was significantly correlated ( $r^2 = 0.893$ ), indicating possible binary mixing of Pb. It should be noted that the Pb isotopic composition of NR-7 was analyzed repeatedly to verify the results. The low  $^{206}\text{Pb}/^{207}\text{Pb}$  and  $^{208}\text{Pb}/^{207}\text{Pb}$  ratios ( $1.1701 \pm 0.0047$  and  $2.5416 \pm 0.0064$ , respectively) of NR-7 were hence considered as outliers. Even though both LJ and NR meander through the center of Guiyu, the results suggest that LJ receives a considerably large input of Pb from a distinctively different source than those feeding NR.



### 3.4. Chemical speciation of trace metals in the sediments

The sequential chemical distribution of Cu, Pb and Zn in the selected sediment samples from LJ and NR is depicted in Fig. 4. As shown in Fig. 4, the chemical distribution of Cu in the sediments from LJ was noticeably different from that in the sediments from NR. In general, Cu in the LJ sediments was dominantly associated with the organic fraction (67.5–87.5%) and slightly with the residual fraction (9.7–19.4%). In the sediments of NR, Cu showed a strong association with both the organic (31.1–50.8%) and residual fractions (37.2–53.0%). In comparison, Cu in the exchangeable, carbonate and Fe–Mn oxides fractions was less significant in the LJ sediments than in the NR sediments. In the LJ sediments, the exchangeable, carbonate and Fe–Mn oxides fractions represented 0.5–1.5, 0.9–7.7 and 0.6–4.9% of total Cu, respectively. In the NR sediments, <0.7, 4.5–12.3 and <7.8% of total Cu were found in the exchangeable, carbonate and Fe–Mn oxides fractions, respectively.

Pb in the sediments from LJ and NR generally exhibited similar distribution patterns where the Fe–Mn oxides fraction was most dominant, ranging from 37.0 to 49.9% and from 30.3 to 49.8%, respectively, followed by the carbonate fraction, ranging from 20.9 to 26.8% and from 20.2 to 36.2%, respectively. The exchangeable, organic and residual fractions accounted for 0.5–1.6, 11.9–14.9 and 15.7–26.0% of total Pb in the LJ sediments, respectively, and 0.3–7.6, 13.7–31.2 and <14.9% of total Pb in the NR sediments, respectively. The sequential chemical distribution of Zn in the sediments collected from LJ and NR was distinctively different. In the LJ sediments, the distribution of Zn showed a relatively even distribution pattern with a decreasing order of association with the Fe–Mn oxides fraction (32.1–40.2%), carbonate (17.9–27.6%), residual (13.3–29.2%), exchangeable (9.5–16.9%) and organic fractions (6.4–9.9%). In the NR sediments, the majority of Zn was found in the residual fraction (53.7–77.9%), followed by the Fe–Mn oxides (10.8–23.6%), carbonate (3.9–12.1%), organic (5.7–8.3%) and exchangeable fractions (0.2–2.7%). In general, Zn in the non-residual fractions was more significant in the LJ sediments than in the NR sediments.

## 4. Discussion

Based on the trace metal concentrations of the sediments, it was evident that the sediments collected from LJ were heavily contaminated with trace metals, particularly Cd, Cu, Ni, Pb and Zn. Whereas, the sediments collected from NR were comparatively uncontaminated with trace metals, except for NR-10. Since sediments

of NR were finer in particle size than LJ, this suggested that particle size did not play a significant role in terms of distribution and dispersion of the contaminants. However, trace metal distribution of the sediments was found to be significantly correlated with TOC, as Pearson correlation coefficients between TOC and trace metals were high (0.791 for Cd, 0.877 for Cu, 0.916 for Ni, 0.925 for Pb, and 0.928 for Zn). Furthermore, Cu, Pb and Zn concentrations of some of the LJ sediments, i.e. LJ-6 and LJ-10, exceeded both the PELs by CCME and the intervention values by the Netherlands Environmental Standards, suggesting the need for further investigation. Since these sediments had a very fine particle size consisting mostly of clay and silt, the sediments could easily be transported as suspended particles, causing trace metal enrichment of the downstream environment. The distribution of Cu, Pb and Zn concentrations of the LJ sediments indicated that these trace metals were possibly discharged into the river at various points and that these metals might be discharged simultaneously. It is, however, unclear why the differences in trace metal concentrations between the two rivers occurred, but it might be a reflection of the effects of different geochemical conditions of the rivers and types of e-waste processing activities and their resultant discharges. One of the possible influential factors was the low pH measured (pH 4.63) in NR. The dumping of strong acids into the river could have lowered the pH of the river and increased metal solubility, hence reducing sorption of trace metals to sediments.

The Pb isotopic composition revealed that the contaminated sediments in LJ received a significant input of non-indigenous Pb with low  $^{206}\text{Pb}/^{207}\text{Pb}$  and  $^{208}\text{Pb}/^{207}\text{Pb}$  ratios. The linear relationship of the  $^{206}\text{Pb}/^{207}\text{Pb}$  and  $^{208}\text{Pb}/^{207}\text{Pb}$  ratios further suggested that Pb in the LJ and NR sediments might have originated from two geological sources, one with high  $^{206}\text{Pb}/^{207}\text{Pb}$  and  $^{208}\text{Pb}/^{207}\text{Pb}$  ratios and the other with low  $^{206}\text{Pb}/^{207}\text{Pb}$  and  $^{208}\text{Pb}/^{207}\text{Pb}$  ratios. Since the Pb-contaminated sediments generally had lower  $^{206}\text{Pb}/^{207}\text{Pb}$  and  $^{208}\text{Pb}/^{207}\text{Pb}$  ratios than the uncontaminated sediments, it was plausible to conclude that the non-indigenous Pb was characterized by low  $^{206}\text{Pb}/^{207}\text{Pb}$  and  $^{208}\text{Pb}/^{207}\text{Pb}$  ratios while indigenous Pb was represented by the high  $^{206}\text{Pb}/^{207}\text{Pb}$  and  $^{208}\text{Pb}/^{207}\text{Pb}$  ratios. The Pb isotopic composition of the uncontaminated sediments was consistent with that of indigenous Pb sources previously reported, where the  $^{206}\text{Pb}/^{207}\text{Pb}$  and  $^{208}\text{Pb}/^{207}\text{Pb}$  ratios were 1.1952 and 2.4815 for uncontaminated soils, respectively, and 1.1993 and 2.4964 for volcanic rock in Foshan, respectively (Zhu et al., 1989 and Zhu et al., 2001).

The sequential chemical distributions of the sediments showed distinctive differences in the mobility, solubility and potential bioavailability of trace metals between LJ and NR sediments. In general, trace metals in the residual fraction remain relatively stable

and inert, and they are not easily released into the mobile and bioavailable phases (Tessier et al., 1979). In contrast, trace metals in the non-residual fractions, including the exchangeable, carbonate, Fe–Mn oxides and organic fractions, may reflect various degrees of reactivity and potential bioavailability. They may also change depending upon the surrounding physical and geochemical conditions (Tessier et al., 1979, Song and Breslin, 1998 and Ryan et al., 2002).

The sequential chemical distribution of Cu, Pb and Zn in the contaminated sediments from LJ showed that a larger percentage of these trace metals resided in the non-residual fractions than NR sediments (see Fig. 4). Generally, trace metals in the non-residual fractions increased in the order of  $Zn < Pb < Cu$  in the LJ sediments and  $Pb < Cu < Zn$  in the NR sediments. As anticipated, Cu in both LJ and NR sediments was dominantly found in the organic fraction indicating that it was strongly affiliated with organic complexes and sulphide (Peng et al., 2004). A particular concern was associated with Cu in the organic fraction of the LJ sediments, as Cu in this fraction of the LJ sediments accounted for 67.5–87.5% of the total Cu, effectively representing as much as 1600 mg Cu/kg. Physical and geochemical changes of LJ, such as removal of organic matter (i.e. reduction of municipal wastewater discharge) and oxidation of anaerobic sediments, could potentially trigger the release of Cu from the sediments, leading to an increase in dissolved Cu concentration in river water (Allen and Hansen, 1996 and Lu and Allen, 2001).

The distribution of Pb in the five fractions appeared to be relatively similar between the LJ and NR sediments, despite their difference in total Pb concentrations. However, it was observed that Pb in the exchangeable fraction of NR sediments was higher than that in the LJ sediments. This might be the effect of low pH in the NR sediments. The six LJ sediments had fairly consistent pH, ranging from 6.05 to 6.63; whereas, pH of NR sediments increased in the order of NR-1 and NR-2 (4.63 and 4.72, respectively) < NR-3 and NR-4 (5.66 and 5.83, respectively) < NR-5 and NR-6 (7.54 and 7.74, respectively). This indicated that the low pH of the sediments resulted in an increase in solubility and potential bioavailability of Pb in the river and that acid dumping could alter the geochemistry of the aquatic system.

Similar to Cu, the chemical distribution of Zn in the LJ sediments differed significantly from the NR sediments. This chemical distribution pattern was consistent with the distribution of total Zn concentrations in the sediments. In general, Zn in the non-residual fractions in the LJ sediments represented approximately 80% of the total Zn, while that in the NR sediments accounted for approximately 22–45%. The large percentages of exchangeable and carbonate fractions of Zn in the LJ

sediments, representing as much as 170 mg Zn/kg, were a concern, as Zn in these fractions could be readily altered into dissolved/soluble forms rendering them mobile and readily bioavailable (Tessier et al., 1979 and Dollar et al., 2001). In the NR sediments, the exchangeable Zn was noticeably lower in the sediments with pH >7 than in those with pH <6. Furthermore, a decreasing trend was observed for Zn in the carbonate and Fe–Mn oxides fractions with increasing pH in the NR sediments. This might also be the effect of pH and possibly a sign of trace metal redistribution in the sediments.

The fact that active aquacultural activities are taking place in the contaminated river of LJ warrants a genuine health concern, as there are still substantial agricultural activities within and immediately surrounding the town on farmlands and in waterways. It is also noteworthy to mention that the rivers studied are part of the irrigation network from which water is extracted for crop irrigation. Long-term use of the water may, in turn, elevate trace metal loadings of the crop and paddy soils, potentially resulting in high levels of uptake of toxic trace metals in rice and other edible food crops (Mueller, 1994).

## 5. Conclusion

This paper offers some scientific evidence of environmental degradation caused by primitive e-waste processing activities in developing countries. In this study, the analytical results showed that Cd, Cu, Ni, Pb and Zn contamination of sediments in Guiyu was serious. Trace metal contamination of the sediments in Guiyu could have significant impacts on the health of the residents and environmental quality of the local and downstream terrestrial and aquatic environment, and further investigations are needed. More importantly, a proactive material management strategy should be taken to minimize e-waste production and make their components more desirable for recycling and reuse.

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# MAP OF GUIYU

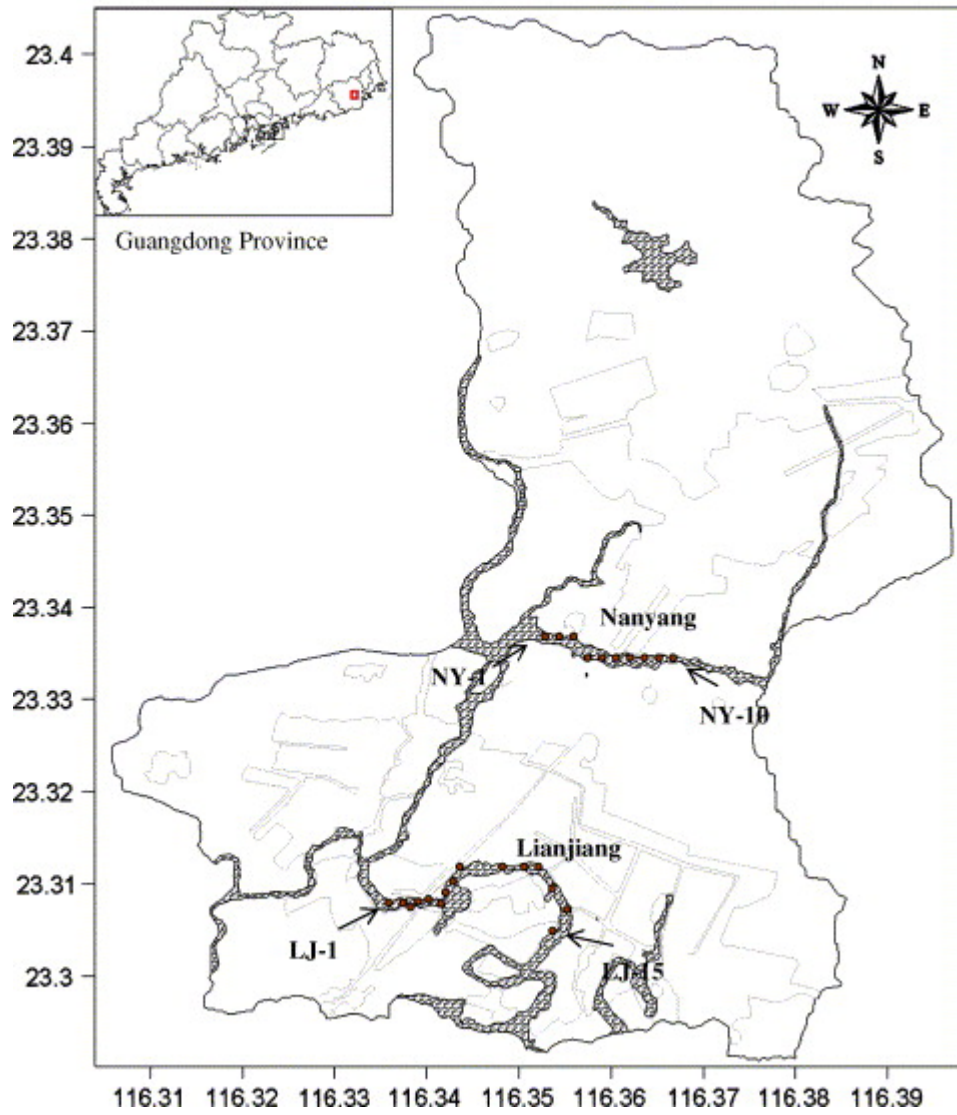


Fig. 1. Sediment sampling locations in Guiyu, Guangdong Province of China.

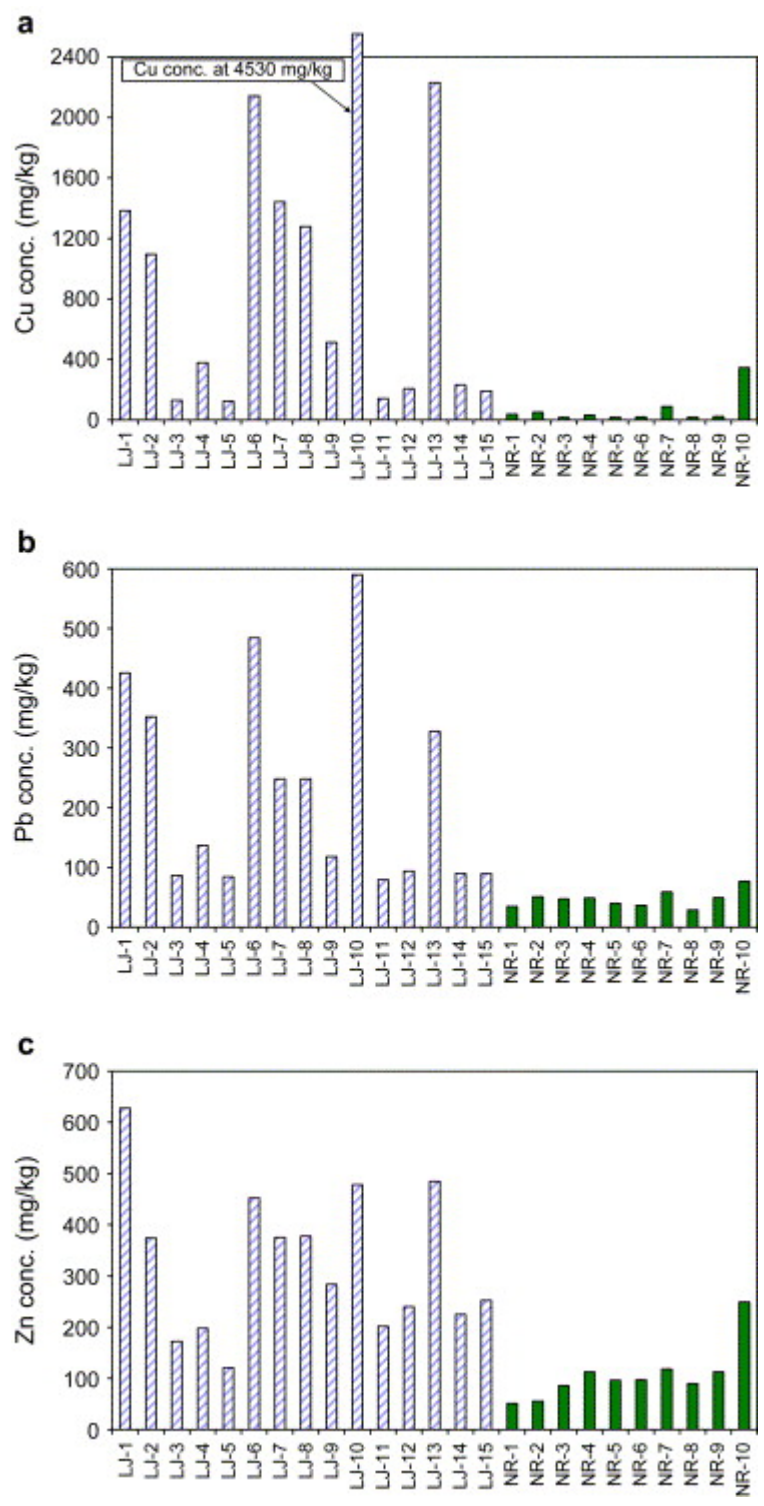


Fig. 2. Distribution of (a) Cu, (b) Pb and (c) Zn in the sediments in Lianjiang (LJ) and Nanyang River (NR).



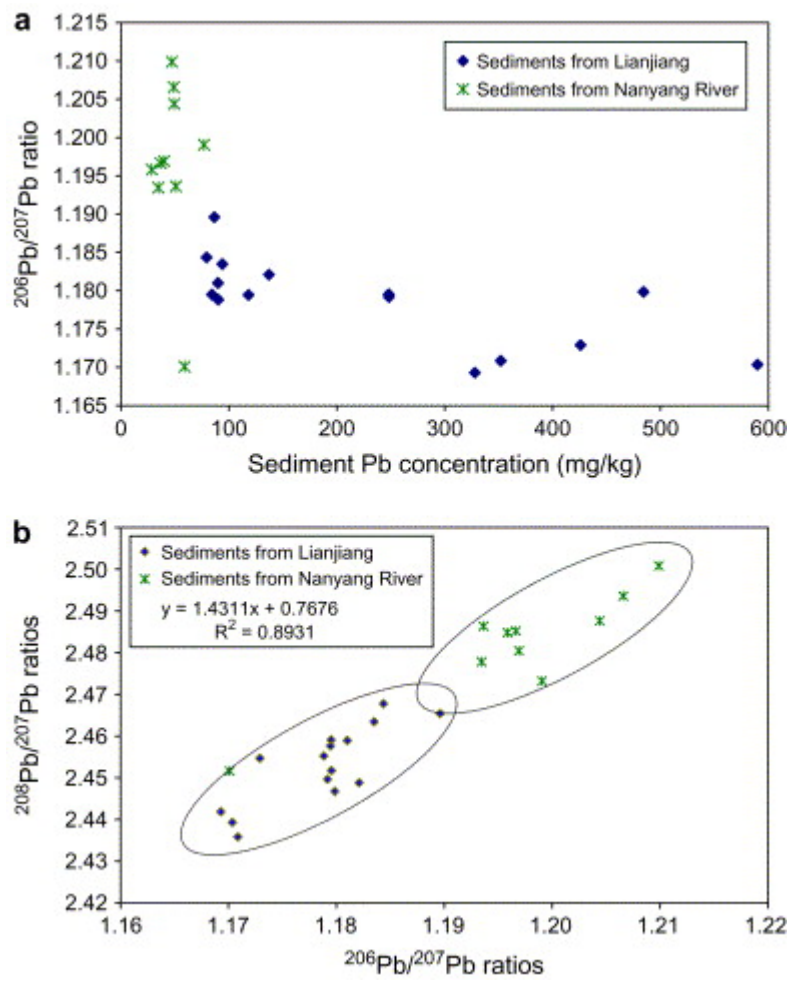


Fig. 3. The distribution of Pb concentrations,  $^{206}\text{Pb}/^{207}\text{Pb}$  and  $^{208}\text{Pb}/^{207}\text{Pb}$  ratios of the sediments from Lianjiang and Nanyang River.

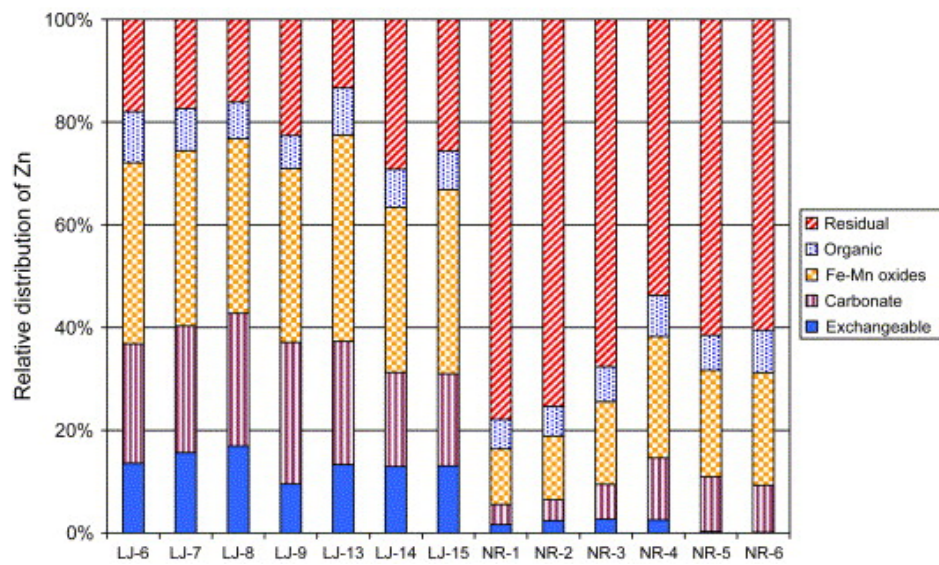
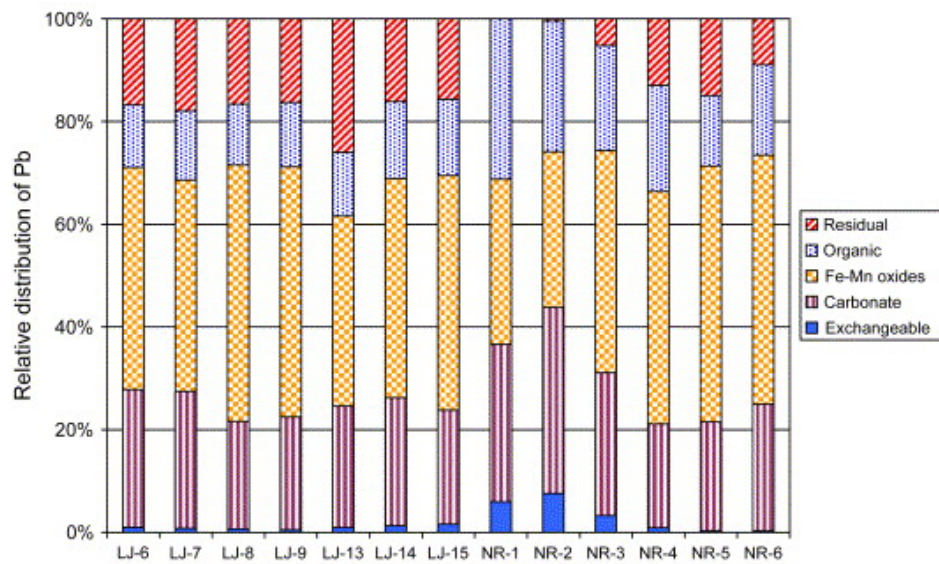
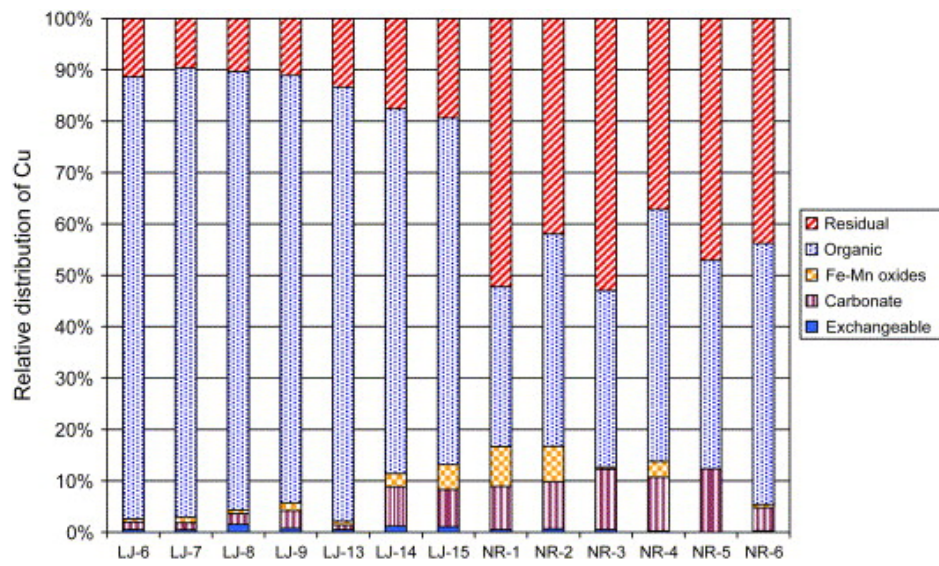


Fig. 4. Sequential chemical distribution of Cu, Pb and Zn of selected sediments.

Table 1  
Physical characteristics of the sediments from Lianjiang and Nanyang River

	Unit	Lianjiang	Nanyang River
No of samples	—	15	10
pH			
Mean ± S.D.	—	6.44 ± 0.41	6.40 ± 1.19
Median	—	6.38	6.50
Range	—	6.03–7.75	4.63–7.74
TOC			
Mean	mg/kg	37.2 ± 20.3	11.3 ± 3.78
Median	mg/kg	33.0	10.2
Range	mg/kg	13.02–81.7	7.35–18.9
Clay (<2 μm)			
Mean ± S.D.	%	5.98 ± 1.10	12.1 ± 6.60
Silt (>2–63 μm)			
Mean ± S.D.	%	55.2 ± 9.15	78.8 ± 8.74
Fine sand (>63–200 μm)			
Mean ± S.D.	%	21.0 ± 4.81	8.37 ± 4.05
Medium sand (>200–800 μm)			
Mean ± S.D.	%	17.8 ± 10.0	0.75 ± 0.57

Table 2  
Trace metal concentrations of sediments from Lianjiang and Nanyang River (mg/kg)

	Lianjiang	Nanyang River	Canada <sup>a</sup>		LJ	NR	Netherlands <sup>b</sup>		LJ	NR
	<i>N</i> = 15	<i>N</i> = 10	ISQG	PEL	<i>N</i> > PEL	<i>N</i> > PEL	Target	Intervention	<i>N</i> > Inter	<i>N</i> > Inter
<b>Cd</b>										
Mean ± S.D.	4.09 ± 3.92	n.a.	0.60	3.50	8	nil	0.8	12	nil	nil
Range	n.d.–10.3	n.d.–0.57								
<b>Co</b>										
Mean ± S.D.	15.5 ± 5.40	12.7 ± 3.63	–	–	n.a.	n.a.	9	240	nil	nil
Range	9.51–24.1	9.10–20.1								
<b>Cu</b>										
Mean ± S.D.	1070 ± 1210	65.1 ± 101	35.7	197	11	1	36	190	12	1
Range	125–4540	17.0–346								
<b>Ni</b>										
Mean ± S.D.	181 ± 156	25.2 ± 10.5	–	–	n.a.	n.a.	35	210	7	nil
Range	26.0–543	12.4–39.8								
<b>Pb</b>										
Mean ± S.D.	230 ± 169	47.3 ± 13.8	35.0	91.3	10	nil	85	530	1	nil
Range	79.5–590	28.6–76.9								
<b>V</b>										
Mean ± S.D.	54.3 ± 5.92	62.6 ± 15.1	–	–	n.a.	n.a.	42	250	nil	nil
Range	41.8–63.5	45.1–89.1								
<b>Zn</b>										
Mean ± S.D.	324 ± 143	107 ± 54.9	123	315	7	nil	140	720	nil	nil
Range	121–628	51.3–249								

n.a., not applicable; n.d., not detectable; ISQG, interim sediment quality guideline; PEL, probable effect level; Target values: levels of trace metals should be achieved; Intervention values: above which remedial actions should be taken.

<sup>a</sup> Canadian Environmental Quality Guidelines.

<sup>b</sup> Netherlands Environmental Quality Guidelines.