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Review

A review on human health consequences of metals exposure to e-waste in China

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ABSTRACT

As the world's the largest dumping ground for e-waste, much of the population in China is exposed to heavy metals due to informal e-waste recycling processes. We reviewed recent studies on body burdens and human health effects of heavy metals from the major e-waste recycling sites in China. The results showed that the residents in the e-waste recycling sites are facing a potential higher daily intake of heavy metals. Moreover, heavy metals had entered subjects' bodies (the collected 5 tissue samples). Additionally, individual exposure to heavy metals in e-waste has also caused negative health outcomes, especially in neonates and children. We also recorded plausible outcomes associated with exposure to e-wast (to heavy metals). A precautionary approach toward exposure, especially in neonates and children, therefore seems warranted.

1. Introduction

With the development of the electronic industry and information technology, a large amount of waste electric and electronic equipment (e-waste) (20–50 million tons per year) is being continually generated worldwide, and has become a serious problem of environmental protection as well as a risk to human health (Duan et al., 2009; Wang et al., 2013). E-waste is a crisis not only of quantity but also of toxic components, such as lead, chromium, cadmium, etc. China, as the world's leading manufacturing country, has become the largest dumping ground for e-waste (Chi et al., 2011; Song et al., 2012a, 2012c, 2013).

Much e-waste recycling in China is a conglomeration of processes carried out in the informal sector (Chan and Wong, 2013; Chen et al., 2009), by a range of legal and publicly accepted, although unregistered, businesses, who give little concern to illegal or clandestinely executed processes that can generate consequences harmful to both the environment and human health. Such processes include applying crude methods to separate substances or material of interest from their original location within the electrical/electronic equipment (Leung et al., 2007; Leung et al., 2013). The operations commonly used in processing e-waste in order to extract precious metals, such as strong acid leaching and the open burning of dismantled components, have led to the release of large quantities of toxic metals and organic pollutants into the surrounding environment (Bi et al., 2010; Birloaga et al., 2013; Fu et al., 2011; Grant et al., 2013; Gullett et al., 2007; Hischier et al., 2005; Leung et al., 2008, 2011; Wong et al., 2007).

E-waste recycling in China, especially informal e-waste recycling, has clearly become a major source of toxic heavy metals (Luo et al., 2011; Tang et al., 2010; Wong et al., 2007). Heavy metals are widely used in the manufacturing of a variety of electronic products, such as lead and cadmium in circuit boards, cadmium in computer batteries, and copper in electrical wiring (Achillas et al., 2013; Song et al., 2012b; Stevels et al., 2013; Zeng and Li, 2013; Zeng et al., 2014). It is well known that heavy metals persist in the environment and lead to poisoning at low concentrations through bioaccumulation in plants and animals or bioconcentration in the food chain (Fu et al., 2008; Luo et al., 2011; Zhang and Hang, 2009). Heavy metals can be absorbed by plants through uptake from the soil, and by animals and humans through food, water, air, soil/dust ingestion and skin contact (Li et al., 2011b; Zhao et al., 2010). Some heavy metals can become more concentrated when people ingest meat, which is higher on the food chain. In humans, lead interferes with behavior and learning abilities; copper results in liver damage; and chronic exposure to cadmium increases the risk of lung cancer and kidney damage (Balakrishnan Ramesh et al., 2007; Bhutta et al., 2011; Chan and Wong, 2013; Esteban-Vasallo et al., 2012; Grant et al., 2013; Yan et al., 2013). Children are particularly susceptible to heavy metal exposure due

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to high gastrointestinal uptake and the permeable blood–brain barrier.

Although many studies have investigated and discussed the environmental pollution of heavy metals from e-waste recycling in China, to the best of our knowledge, no systematic reviews specifically focused on body burdens, and human health effects of heavy metals have been performed. Considering the above-mentioned situation, this study reviews the current state of knowledge on heavy metal human exposure to heavy metals from e-waste in China, with almost all the representative data now available on exposure sources, human tissue markers, and human health in China. The main objectives of this study were to provide comprehensive information on the current human health effects of heavy metals and to assess the evidence for the association between such heavy metals exposure and the human body burden (or human health) in China. Fig. 1 presents the e-waste exposure areas and the control areas referred to in this paper. As shown in Fig. 1, most studies mainly focused on the southeast region of China, because e-waste recycling processed in China (especially the informal sectors) is mainly limited to those regions, e.g. Guiyu, Taizhou, and Qingyuan.

2. Review methods

We systematically searched the electronic databases (Web of Knowledge, Science Direct, Google Scholar, CNKI (database of Chinese journal)) for the search terms (e-waste, electronic waste, WEEE, heavy metals, body burden, health, and exposure) from Jun 1 2013 to Sep 2014. Of the 31 full-text articles assessed for eligibility, we excluded the studies reporting results in reviews, letters to the editor, and abstracts, and those that did not report an outcome related to heavy metal effects from e-waste, resulting in these published studies that met the searching criteria. To focus our study, we did not consider exposure to other pollutants (PBDEs, PCBs, and PCDD/Fs) or studies in other countries (such as India and African nations). Our search was not restricted to the English language, nor by any other means. Relevant articles published in languages other than English, especially Chinese, were translated. A data extraction sheet was pilot tested and revised to include: publication details, study characteristics (period, location, and sampling size), exposure and outcome measures, and study outcome.

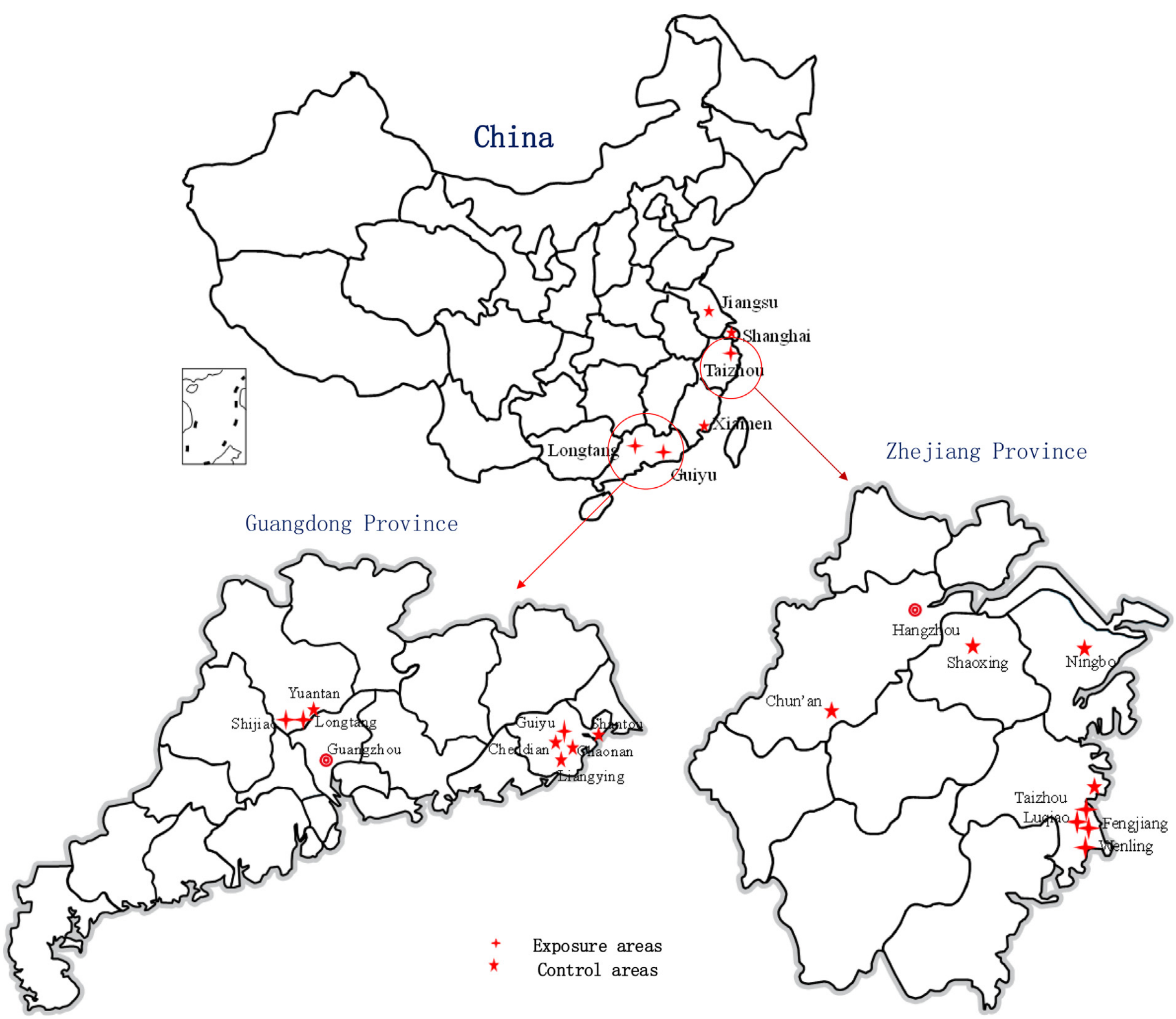

Fig. 1. E-waste exposure and control areas referenced, in China.

Table 1
Exposure sources of heavy metals from e-waste.

Location	Sampling time	Exposure ways	Exposure groups	Concentrations	References
Taizhou	2005.11	Rice	Residents	As: 0.8, Cd:0.7, Hg:0.1, Pb: 3.7 μg/(day·kg bw)	(Fu et al., 2008)
Longtang	2007.9	Vegetables	Residents	Cd: 1.36 μg/(day·kg bw); Cu: 10.6 μg/(day·kg bw); Pb: 3.00 μg/(day·kg bw); Zn: 69.8 μg/(day·kg bw)	(Luo et al., 2011)
Rural region in South China	–	Food (vegetables, rice, pork, chicken and fish); House dust; Groundwater	Residents (adults, and children)	Adults: Cd: 5.28; Pb:54.2; Cu:155; Zn:228; Ni: 15.7 μg/(day·kg bw); Children: Cd: 8.95; Pb:81.2; Cu:316; Zn:502; Ni: 26 μg/(day·kg bw);	(Zheng et al., 2013b)
Guiyu	2004.12	Dust	Workers and children	Workers: Pb: 175.7; Cu: 13.2; Cd: 0.043; Cr: 0.11 μg/(day·kg bw); Children: Pb: 1421; Cu: 106.8; Cd: 0.34; Cr: 0.93 μg/(day·kg bw)	(Leung et al., 2008)
Jiangsu (Formal)	–	Air	Workers	ADD_{ing}: Cr: 49.1; Cu 20,900; Cd 16; Pb: 4930 ng/(day·kg bw); ADD_{inh}: Cr: 0.0055; Cu: 2.33; Cd: 0.0018; Pb: 0.013 ng/(day·kg bw); ADD_{derm}: Cr: 0.22; Cu: 94; Cd: 0.072; Pb: 22.2 ng/(day·kg bw)	(Xue et al., 2012)
Shanghai (Formal)	–	Air (PM2.5)	Workers	Mechanical workshop: ADD_{ing}: Cr: 1371; Cu: 2652; Cd: 37.7; Pb: 23,345 ng/(day·kg bw); ADD_{inh}: Cr: 0.15; Cu: 0.48; Cd: 0.0032; Pb: 2.61 ng/(day·kg bw); ADD_{derm}: Cr: 6.18; Cu: 1.932; Cd: 0.17; Pb:105.53 ng/(day·kg bw); Manual dismantling workshop: ADD_{ing}: Cr: 1083; Cu: 592; Cd: 107; Pb: 7910 ng/(day·kg bw); ADD_{inh}: Cr: 0.12; Cu: 0.15; Cd: 0.012 Pb: 2.39 ng/(day·kg bw); ADD_{derm}: Cr:4.89; Cu: 0.62; Cd: 0.48; Pb:35.65 ng/(day·kg bw)	(Fang et al., 2013)

3. Results

3.1. Exposure sources and health risk

3.1.1. Exposure sources

In general, exposure to the hazardous components of e-waste can occur through three ways: inhalation from air, dietary intake, soil/dust ingestion and skin contact. In addition to direct occupational (formal or informal) exposure, people can come in contact with e-waste materials, and associated pollutants, through contact with surrounding soil, dust, air, and water, and through food sources. Hence, in order to understand how or how much heavy metals were transferred into the human body, it will be very essential to investigate the exposure sources and average daily dose of heavy metals from e-waste.

Exposure sources to heavy metals from e-waste are presented in Table 1. Fu et al., 2008 evaluated Taizhou residents' dietary intakes of heavy metals from rice. Comparing the tolerable daily intakes given by FAO/WHO (Organization, 2011) with the mean estimated daily intakes, Pb daily intake through rice consumption in this area was 3.7 μg/(day·kg bw), which already exceeded the FAO tolerable daily intake (3.6 μg/day·kg bw), and the Cd daily intake (0.7 μg/day·kg bw) through rice reached 70% of the total tolerable daily intake (1 μg/day·kg bw). Luo et al., 2011 calculated residents' ingesting heavy metals by consuming vegetables grown around the sampling area in Longtang town. The daily intake level of Cd (1.36 μg/(day·kg bw)) exceeded about 2.72 time of the oral reference dose (0.5 μg/(day·kg bw)). Compared with above two studies, Zheng et al., 2013b investigated more sources, including food (vegetables, rice, pork, chicken and fish), house dust, and groundwater, to estimate the daily intakes. Without any exception, the daily intakes were higher, with more serious health risk, especially the children.

Compared with the residents in the e-waste recycling areas, the workers in e-waste recycling workshops were faced with more serious potential health risks. According to Leung et al., 2008, for a printed circuit board recycling worker in Guiyu, the Pb inhalation from air 175.7 μg/(day·kg bw), indicating that the estimated oral average daily dose (ADD) much exceeded the “safe” oral Pb reference dose. Furthermore, the potential health risk to children at all locations was 8 times that for workers, because of the children's smaller body size and the higher ingestion rate used (200 mg/kg/day) in estimating the risk (Leung et al., 2008). Compared with the informal e-waste recycling sites in Guiyu, the workers from formal e-waste recycling sites in Jiangsu and Shanghai (Fang et al., 2013; Xue et al., 2012) had lower potential health risks from heavy metals, especially for Pb intake (excess reference dose) (Konz et al., 1989). It was also found that the workers' average daily dose contracted through ingestion (ADD_{ing}) was the highest, accounting for more than 90% of the total intake, followed by the average daily doses absorbed through skin (ADD_{derm}) and through inhalation (ADD_{inh}). In addition, according to Fang et al., 2013, the workers' ADD of four heavy metals (Cr, Cu, Cd, and Pb) in a mechanical workshop were higher than those in the manual dismantling workshop, possibly because the mechanical operation can generate more particles and dust.

3.1.2. Health risk of heavy metals

The hazards of heavy metals on human health are well known. Through the estimated daily intake, some studies evaluated the potential health risks from heavy metals. The model of health risk assessment from the U.S. EPA was often applied to evaluate the hazard quotients (HQ) of heavy metals for each non-carcinogenic metal (Xue et al., 2012). Except for the non-carcinogenic risks, the carcinogenic risks were also assessed in some studies.

The health risk of heavy metals related to non-carcinogenic and carcinogenic were given in Table 2. Four studies including two formal and two informal e-waste exposure. As listed in Table 2, even in the two formal e-waste recycling sites, the workers were also facing on the potential health risk from heavy metals. In the formal production Line for Recycling Waste Printed Circuit Boards in Jiangsu province (Xue et al., 2012), the workers' non-cancer effects might be possible for Pb (HQ = 1.45), and non-cancer effects are unlikely for Cr, Cu, and Cd. The carcinogenic risks for Cr and Cd are 3.29×10^{-8} and 1.61×10^{-9}, respectively. It indicates that cancer risks on workers are relatively light in the workshop (the range of threshold value was 10^{-6}). For mechanical workshop and manual dismantling workshop of e-waste in Shanghai (Fang et al., 2013), though there was also the potential noncancerous risk of Pb > 1, the cancer risk of Cr exceeded the threshold obviously, therefore it may have a cancer risk on the workers. The lifetime cancer risk of four targeted metals was Cr > Ni > Pb > Cd, which could be proven in all monitoring samples. Comparing with the formal e-waste recycling sites, the residents' non-cancer and cancer risk of informal recycling sites were more serious, especially for the children (Zheng et al., 2013b; Zhang et al., 2014). As reported by Zheng et al., 2013b, the residents' non-cancer effects might be very possible for Cd, Pb, Cu, Zn (all >1), and there was an obvious cancer risk of Pb (Adults:Pb:5×10^{-4}; Children: Pb: 1×10^{-3}). Noted that all the children will be facing with more serious health risk than the Adults.

3.2. Human body burden of heavy metals

The biomonitoring of heavy metal concentrations in the environmental matrices—for example, air, soil, sediment, dust, and food—has revealed the presence and the levels of such pollution in the e-waste processing sites in China. However, the presence of the pollutant in the environment does not in itself determine the intensity of human exposure to heavy metals from the e-waste recycling processes (Esteban and Castaño, 2009; Graber et al., 2010); such determination requires specific studies. Through investigating the human body burden of heavy metals, it will be very helpful to understand the potential human health effects. The current studies on the human body burden of heavy metals have included 5 types of human tissues: placenta, umbilical cord blood, blood and serum, hair, and urine, as shown in Table 3.

3.2.1. Placenta

Three studies have examined the human body exposure to e-waste in placentas collected after childbirth; the results are shown in Table 3. As for the heavy metal levels in placentas, although all three studies investigated the human body impacts from the same exposure sites (Guiyu), the results were not consistent. In two studies (Li et al., 2011c; Zhang et al., 2011a), the Cd levels in placentas from Guiyu were significantly higher than those from Chaonan ($p < 0.01$ and $p < 0.001$), whereas there was no significant correlation in Guo et al., 2010. Pb levels also showed different significant correlations (Guo et al., 2010; Zhang et al., 2011a). The total Pb value in placentas in Guiyu was about twice that of the reference group in Chaonan (301.43 vs. 165.82 ng/g), and was significant ($p = 0.010$) (Chan et al., 2007). According to Zhang et al., 2011a, the median Cd level was also higher, but not significantly so, in Guiyu, compared to Shantou (521.01 vs. 273.24 ng/g, $P = 0.299$). Also in this study, a negative but not significant correlation of Ni level was found, between Guiyu and Chaona ($p = 0.000$).

3.2.2. Umbilical cord blood

Only a few studies have investigated the pollutant levels in umbilical cord blood (UCB) due to e-waste exposure. All the heavy metal levels in UCB from the exposure areas, however, were higher than those from the control areas (Table 3).

Among the 423 mothers recruited from Guiyu and Chaonan, in 2004/2005, 2006, and 2007 (Li et al., 2011c), the median Cd level in UCB was significantly higher for Guiyu neonates than for controls (3.61 vs. 1.25 μg/L, $p < 0.01$), with 25.61% of Guiyu subjects exhibiting a median Cd level that exceeded the safety limit defined by the World Health Organization (5 μg/L), as compared with 14.18% of control neonates ($p < 0.01$). Similarly, according to Ni et al., 2013b, UCB cadmium concentrations in neonates in Guiyu were also significantly higher than those in Shantou (median 2.50 vs. 0.33 ng/mL, $p < 0.001$). For the Pb level in UCB, the three studies (Li et al., 2008b; Ni et al., 2013b; Xu et al., 2012) obtained similar results: compared with the control group, neonates in Guiyu had significantly higher levels of lead in UCB (Pb > 100 μg/L, $p < 0.01$).

3.2.3. Blood

The assessment of human exposure to e-waste has increasingly relied on human blood as a biomarker, and many studies of the human body burden caused by e-waste exposure have also concentrated primarily on heavy metal levels in blood and serum, as shown in Table 3.

For assessing the body burden in human blood, the Pb level has attracted the most attention, and 12 studies have investigated blood lead levels in the human body in order to understand e-waste exposure. According to the diagnostic blood-lead-level criteria for children, defined by the U.S. Centers for Disease Control (Control, 1991), children with a blood lead level (BLLs) ≥100 μg/L are considered to have an elevated blood lead level. Aside from two study that included no statistical analysis, all the results showed that the blood lead levels in children in the exposure areas were significantly higher than those in the control areas (all $p < 0.01$), and most levels exceeded the control standards, indicating that children are more sensitive to e-waste exposure, and have higher potential health effects compared with adults. In addition, Zheng et al., 2008 found that in Guiyu the BLLs, as well as the proportions of BLLs greater than 100 μg/L, increased with the children's age, with older children tending to have higher BLLs than younger ones. It also found that the BLLs of children aged 5, 6, and 7 years of age in Guiyu were significantly higher than those in Chendian (all $p < 0.01$), but no significant differences could be found between males and females in Guiyu and Chendian, results consistent with Liu et al., 2011. Children in Guiyu and Taizhou had higher concentrations of lead in their blood than those living in towns with no e-waste recycling, and this result was further correlated with the location of residence in Guiyu (with particularly high levels in those living within 50 m of an e-waste workshop), the use of the home as a recycling workshop, and the gnawing of toys by children. Yang et al., 2013b also found that as the increasing age, the BLLs of occupational males in Taizhou would increase (group aged under 31: 98.55 μg/L; group aged 30–45: 100.23 μg/L; group aged 46–60: 100.23 μg/L), which might be explained by the time spent to proceed E-waste pickling dismantling.

Four studies analyzed the connection between blood Cd levels and e-waste exposure, but results were not consistent. First, there were large differences among the blood Cd levels: 0.69 μg/L, 1.287 (or 1.34) μg/L, and 7.91 μg/L, for Guiyu, Southeast China, and Taizhou, respectively (Wang et al., 2011b; Yang et al., 2013a; Zhang et al., 2012), indicating that Taizhou children were faced with a more serious Cd body burden compared with those in Guiyu or Southeast China. Secondly, the blood Cd level increased significantly in the exposed group in Taizhou compared with the control group ($p < 0.05$). By contrast, in Southeast China (Wang et al., 2012a), the control group had a higher blood Cd level, and similar results were found in the blood Cu level (Wang et al., 2011b; Zhang et al., 2012). However, the author provided no explanation for this result. The blood levels of other heavy metals levels (Cr, Hg, Ni, Mn, Fe) are also presented in Table 3, and there were the similar results with the blood Pb and Cd levels.

3.2.4. Hair

Heavy metals can enter the hair by various pathways, such as incorporation from the blood and deep skin layers during hair shaft formation, deposition by diffusion from sweat or sebum secretions, and the external environment (Pragst and Balikova, 2006). Hair has therefore been identified as a suitable indicator to reflect short- and long-term exposure to heavy metals.

In two studies (Table 3), heavy metal levels in human hair were investigated to understand the human body burden related to e-waste exposure. In one study (Zheng et al., 2011), levels of five heavy metals were found, in the following order: Zn > Pb > Cu > Cd > Ni, with the highest levels found in occupationally exposed workers; the levels of Cd, Pb, and Cu were significantly higher in residents from the e-waste recycling area than in the control area (all $p < 0.01$); elevated Cd, Pb, and Cu levels in hair from the e-waste recycling area, and significant positive correlations between them, indicated that these metals were likely to have originated from participation in e-waste recycling activities. Wang et al., 2009 also found a similar ranking: Pb > Cu ≥ Mn > Cr > Ni > Cd > As, and Cu, Pb, and Cd levels were also found to be higher compared to those in hair samples from control areas. These studies show that human hair can be a useful biomarker for assessing the extent of heavy metal exposure to workers and residents in areas with intensive e-waste recycling activities.

3.2.5. Urine

The level of heavy metals in human urine could be associated with a level of body burden at a particular time, and is an important indicator in health risk analysis. Three studies have investigated heavy metal levels in human urine and their correlations with e-waste exposure (Table 3).

Wang et al., 2011a investigated the impacts of heavy metals in human urine in 349 people from e-waste recycling sites and 118 people from a green plantation. The levels of urinary Cd in both occupational dismantling workers (median1.09 μg/L) and non-occupational-dismantling persons (median 0.75 μg/L) were significantly higher than in the control group (median 0.41 μg/L) ($p < 0.01$ and $P < 0.05$). Further analysis of correlations between urinary heavy metal levels and exposure factors in the exposed group revealed a positive relationship between the duration of dismantling activity and the level of Pb in human urine ($p < 0.05$). According to Wang et al., 2011b, the Cu and Pb levels in Southeast China were also higher than in the reference site. By contrast, urinary cadmium levels were higher in the controls (median: 2 μg/g creatinine) than in the exposed individuals (median: 0.001 μg/mg creatinine, $p < 0.05$). Zhang et al., 2007 obtained similar results (Cd: 1.04 vs. 1.98 μg/g creatinine, $p < 0.01$).

3.3. Health effects of heavy metals

Findings of the studies (Table 4) showed correlations between exposure to heavy metals from e-waste and health outcomes, mainly focusing on neonate's health, children's health, and changes in cellular expression. Outcomes were reported from Guiyu and Taizhou in Southeast China, and mainly included the informal exposure routes.

3.3.1. Neonate's health

Pregnant women's exposure to heavy metals from e-waste has resulted in various negative effects, with increases in spontaneous abortions, stillbirths, premature births, reduced birth weights and infant lengths, Apgar scores, neonatal behavioral neurological assessment (NBNA) scores, and etc.

As shown in Table 4, all the studies investigated the adverse outcomes caused by heavy metal exposure in Guiyu town. According to Li et al., 2008a, Li et al., 2008b, and Guo et al., 2010, there were no significant differences between the adverse birth outcomes (birth weights, infant lengths, and Apgar scores) and the heavy metal exposure. In detail, all the 3 studies showed that the Pb levels in UCB and placentas in Guiyu were significantly higher than those of control groups, but no significant differences in reproductive health between the exposure areas and control areas were found. Not consistent with the above two studies, Xu et al., 2012 reported that Guiyu births showed significantly higher rates of adverse birth outcomes including stillbirth (4.72% vs. 1.03%), low birth weight (6.12% vs. 4.12%), low birth weight (3.40% vs. 1.57%), and lower Apgar scores (9.6 vs. 9.9) and mean birth weight (3168 g vs. 3258 g) than did births from the control site, all $p < 0.01$. In addition, as reported by Li et al., 2008a and Li et al., 2008b, there was a statistically significant difference in NBNA scores between the Guiyu group and the control group by *t* test ($p = 0.043$).

3.3.2. Children's health

Five studies have examined the effects of heavy metals (Pb, Cd, Ni, and Mn) from heavy metal exposure on children's health, focusing mainly on the intelligence quotient (IQ), forced vital capacity (FVC), weight and height, and temperament.

Wang et al., 2012a reported that no significant differences in IQ were observed between Luqiao and Chun'an. According to Han et al., 2007, children's blood lead levels in the exposure groups were significantly higher than those of the control group, and the mean IQ in the exposure area (in children aged from 3 to 4) was also significantly lower than that of control area at the same age (10.24 vs. 12.92, $p < 0.05$). But the intelligence levels of children aged 5 to 6 had no statistically significant difference between two areas. Zheng et al., 2013a evaluated the damage caused by chromium, nickel and manganese exposure on lung function in 144 school children (aged 8–13). The only significant difference between Guiyu and Liangying with regard to lung function was a decreased forced vital capacity (FVC) in 8-to-9-year-old boys (1859 vs. 2,121 ml, $p = 0.003$). Regression analysis showed significant negative correlations between blood chromium levels and FVC in 11- and 13-year-olds ($p = 0.018$, and $p = 0.027$, respectively), and serum nickel levels in 10-year-olds ($P = 0.035$). Huo et al., 2007 found that although children living in Guiyu had significantly higher blood lead levels compared with those living in Chendian ($p < 0.01$), as for physical indexes (weight, height, Head circumference, and Chest circumference), there was no significant difference between Guiyu and Chendian. Liu et al., 2011 evaluated the dose-dependent effects of lead exposure on temperament alterations in 303 children (3–7 years old) from Guiyu and Chendian. Significant differences of mean scores in activity level (4.53 vs. 4.18), approach-withdrawal (4.62 vs. 4.31), and adaptability (4.96 vs.4.67) were found between Guiyu and Chendian children (all $p < 0.01$). Xu et al. (Xu et al., 2013) also investigated children' health effects of Cr in Guiyu and Chendian, but no significant differences for the height, weight, head circumference, and chest circumference of children were found.

3.3.3. Adult's health

Most studies pay more attention on the neonate and children's health effects of heavy metals. We only found one study (Yang et al., 2013b) to assess the sex hormone levels of FSH, LH and T of 187 occupational males in Taizhou. FSH and LH levels increased with the age while the T levels decreased with the age instead. There was a significant correlation between the FSH and LH levels and wearing masks. However, the author didn't analyze the correlation between the sex hormone levels and Pb levels.

3.3.4. Changes in cellular expression

Six studies have been identified that evaluated changes in cellular expression caused by exposure to heavy metals from e-waste pollution, concentrated mainly on Metallothionein (MT), the

Table 2
Health risk assessment of heavy metals.

Location	Exposure groups	Sampling sites	Pollutants	Outcomes	References
Jiangsu (formal)	Workers	Production line for recycling waste printed circuit boards	Cr, Cu, Cd, and Pb	Non-cancer risk (HQ): Cr: 0.0203; Cu: 0.53; Cd: 0.023; Pb: 1.45 Cancer risk: Cr: 3.29×10^{-8}; Cd: 1.61×10^{-9}	Xue et al., 2012
Shanghai (formal)	Workers	Mechanical workshop, and manual dismantling workshop	Cr, Ni, Cu, Cd, and Pb	Mechanical workshop: Non-cancer risk (HQ): Cr: 0.556; Ni: 0.043; Cu: 0.068; Cd: 0.055; Pb: 6.88 Cancer risk: Cr: 3.45×10^{-4}; Ni: 9.43×10^{-6}; Cd: 1.52×10^{-6}; Pb: 5.61×10^{-6} manual dismantling workshop: Non-cancer risk (HQ): Cr: 0.448; Ni: 0.067 Cu: 0.015; Cd: 0.155; Pb: 2.33 Cancer risk: Cr: 1.36×10^{-4}; Ni: 6.56×10^{-6}; Cd: 2.03×10^{-6}; Pb: 9.97×10^{-7}	Fang et al., 2013
Rural region in South China	Residents (adults, and children)	House dust, food, water	Cd, Pb, Cu, Zn, Ni	Non-cancer risk (HQ): Adults: Cd: 5.38; Pb:15.42; Cu:2.38; Zn:4.31; Ni: 0.79; Children: Cd: 8.92; Pb:33.24; Cu:5.54; Zn:6.87; Ni: 1.31 Cancer risk: Adults:Pb:5×10^{-4} Children: Pb: 1×10^{-3}	(Zheng et al., 2013b)
Taizhou	Residents (adults, and children)	Abandoned e-waste recycling plant	Pb, Cd, Cr, Zn, Hg, and Cu	Non-cancer risk (HQ): Adult: Pb, Cd, Hg, and Cu > 10; Zn:6.4; Cr < 1; Children: all >10	(Zhang et al., 2014)

S100 group of calcium-binding proteins, erythrocyte superoxide dismutase (SOD), GSH-Px, the percentage of helper/inducer T lymphocytes (CD4+), and hemoglobin, and DNA.

Higher placental MT expression was found in neonates from Guiyu when compared to those from the control city of Chaonan (Li et al., 2011c). MT expression was 67.00% in Guiyu compared to 32.69% in Chaonan ($P < 0.01$), and was significantly correlated with cord blood and placental Cd levels ($p < 0.01$, and $p = 0.00$, respectively). Zhang et al., 2011a found significant down regulation of S100P proteins, higher MT expression, and higher placental cadmium levels in mothers exposed to e-waste from Guiyu compared to non-exposed mothers from the control town, Shantou. Mean relative mRNA and protein S100P levels were significantly lower in Guiyu compared to Shantou (0.175 vs. 1.462, $p < 0.001$, and 0.026 vs.0.032, $p = 0.045$, respectively); and MT expression was significantly higher in Guiyu than in Shantou (0.051 vs. 0.035, $p = 0.003$), both of which indicated possible exposure to metals, including cadmium. In Zhejiang Province, China, Zhang et al., 2012 reported that the activities of SOD, GSH-Px, and CD4+ in an exposed group were significantly less than those in the contro1group (median 12,401.6 vs. 14,076.6 U/gHb, 57.6 vs. 71.6 nmol/mgHb, 32.7% vs. 37.8%, all $p < 0.05$, respectively). These effects were mainly related to the increase of Cd, Cr and Pb levels in peripheral venous blood. According to Xu et al., 2006, blood lead levels were investigated as the main chemical agent associated with hemoglobin levels. Although the hemoglobin levels of the children with lead poisoning in Guiyu were lower than those of the children with light lead levels, there were no significant associations. Wang et al., 2011b examined the effects of lead exposure on the frequency of MNBNC in e-waste recycling workers in southern China. The exposed group had higher MNBNC frequencies (median: 4.0‰, 1st/3rd quartiles: 2.0–7.0) compared with the controls (median: 1.0‰, 1st/3rd quartiles: 0.0–2.0). MNBNC frequencies and blood lead levels were positively correlated ($r = 0.254$, $p < 0.01$). However, no significant associations were found between Cu and Cd, and MNBNC frequency. In the e-waste recycling town of Guiyu, significant differences in lymphocytic DNA damage in neonates were found compared to the neighboring town Chaonan (Li et al., 2008a). Guiyu neonates had greater DNA damage with significantly higher ($p < 0.01$) injury rates (33.20%) and lengths of tails (4.49 ± 1.92) in the comet assay compared to neonates from Chaonan (10.70% and 2.09 ± 0.65, respectively). Significant correlations were found between blood chromium levels and DNA damage in all of the study neonates (cell populations $r_s = 0.95$, $p < 0.01$, and length of tail $r_s = 0.89$, $p = 0.00$).

4. Discussion

The present study systematically examines the human body burden and health effects of heavy metals from e-waste recycling in China. We assessed the exposure routes and risk, human body and human health effects of heavy metals to understand the evidence of causality between exposure to heavy metals from e-waste and human health outcomes.

When the heavy metals entered the human body through the three route (inhalation from air, dietary intake, soil/dust ingestion and skin contact), there will be the serious body burden. The relevant results proved that the heavy metals have been accumulated in the 5 human tissues (5 tissue: placenta, umbilical cord blood, blood and serum, hair, and urine). Finally, the human body burden of heavy metals would appear on the human health, and cause all kinds of diseases—e.g. cancers, mental health and neurodevelopment disorders, thyroid dysfunction, and general physical health deterioration (DNAdamage and effects on gene expression).

Although many studies have estimated the potential daily intakes of the heavy metals in e-waste recycling sites, it should be noted that the use of data generated in these surveys to estimate dietary exposure, inhalation, soil/dust ingestion and dermal exposure would likely overestimate the actual exposure. There are many factors that can affect the daily intake. First, for dietary intake, different eating habits between areas influence the dietary intake (Chan et al., 2013; Luo et al., 2009; Qin et al., 2011). In addition, food products produced in other sites can also be transported to the local market. Such imports would have lower concentrations, and therefore the level of dietary exposure would be overestimated if they were omitted from the calculation. Second, inhalation exposure could also be overestimated to some extent, because not all the concentrations of particles (i.e. TSP) would be fully available for human inhalation (Li et al., 2007), since many effective measures

Table 3
Human body burden of heavy metals from e-waste.

Pollutants	Sampling time	Exposure groups	Sampling sites	Human body burden	References
Placenta					
Pb, Cd, Cr, Ni	2008.10–2009.5	220 mother–infant pairs (101:119)	Guiyu vs. Chaonan	Pb: 301.43 vs. 165.82 ng/g, $p = 0.010$; Cd: 108.75 vs. 104.15 ng/g; Cr: 234.31 vs. 228.40 ng/g; Ni: 7.64 vs. 14.30 ng/g, $p = 0.000$.	(Guo et al., 2010)
Cd	2006	423 mother–infant pairs (289:134)	Guiyu vs. Chaonan	170 ± 48 0 vs. 100 ± 110 ng/L, $p < 0.01$	(Li et al., 2011c)
Cd, Pb	2008.10–2009.6	105 mother–infant pairs (55:50)	Guiyu vs. Shantou	Cd: 83.99 vs. 51.59 ng/g, $P < 0.001$; Pb: (521.01 vs. 273.24 ng/g, $P = 0.299$)	(Zhang et al., 2011a)
Umbilical cord blood					
Cd	from 2004/2005 to 2007	423 mother–infant pairs (289:134)	Guiyu vs. Chaonan	2004/2005: 4.08 vs. 4.26 μg/L; 2006: 4.17 vs. 1.46 μg/L, $p < 0.01$; 2007: 0.99 vs. 0.63 μg/L, $p < 0.01$; Total: 3.61 vs. 1.25 μg/L, $p < 0.01$	(Li et al., 2011c)
Pb, Cd, Cr, Ni	2012.3–2013.1	201 mother–infant pairs (126:75)	Guiyu vs. Shantou	Pb: 110.45 vs. 57.31 μg/L, $p < 0.001$; Cd: 2.50 vs. 0.33 μg/L, $p < 0.001$; Cr: 27.52 vs. 26.42 μg/L; Ni: 8.63vs. 9.09 μg/L	(Ni et al., 2013b)
Pb	2006	152 mother–infant pairs	Guiyu vs. control area	113.28 vs. 60.43 μg/L, $p = 0.00$	(Li et al., 2008b)
Pb	2001–2008; 2004–2009	531 mother–infant pairs	Guiyu vs. Xiamen	Pb: 107.8 vs. 22.5 μg/L, $p < 0.01$	(Xu et al., 2012)
Blood					
Pb and Cd	–	246 children (aged 3–8)	Only Guiyu	Pb: 73.0 (43.3–154.3) (μg/L); Cd: 0.69 (0.36–1.45) (μg/L); No statistical analysis.	(Yang et al., 2013a)
Pb, Cu, Cd	–	104 persons (48:56)	Southeast China	Pb: 114.49 vs. 91.04 μg/L, $p < 0.01$; Cu: 763.000 vs. 841.500 μg/L; Cd: 1.287 vs. 1.840 μg/L.	(Wang et al., 2011b)
Cd, Pb, Cr, Hg	–	76 workers and residents (40:36)	Taizhou vs. control area	Cd: 7.91 vs. 5.63 μg/L, $p < 0.01$; Pb: 150.63 vs. 84.37 μg/L; Cr: 19.58 vs. 23.48 μg/L, $p < 0.05$; Hg: 3.28 vs. 4.19 μg/L;	(Zhang et al., 2012)
Pb	2004	226 children (165:61)	Guiyu vs. Chendian	153.0 ± 57.9 vs. 99.4 ± 40.5 μg/L, $p < 0.01$	(Huo et al., 2007)
Pb	2006	278 children (aged 1–7) (154:124)	Guiyu vs. Chendian	131.7 ± 59.8 vs. 100.4 ± 48.5 μg/L, $p < 0.01$	(Zheng et al., 2008)
Pb	2008.1–2	303 children (aged 3–7) (153:150)	Guiyu vs. Chendian	144.3 ± 69.3 vs. 87.2 ± 43.4 μg/L, $p < 0.01$	(Liu et al., 2011)
Pb	2004.9–11	226 children (aged 1–6) (165:61)	Guiyu vs. Chendian	153.0 ± 57.9 vs. 99.4 ± 40.5 μg/L, $p < 0.01$	(Xu et al., 2006)
Pb	2006	136 children (aged 3–6) (85:51)	Guiyu vs. Chendian	117.8 vs. 89.3 μg/L, $p < 0.01$	(Han et al., 2007)
Pb	–	226 children (aged 1–6) (165:61)	Guiyu vs. Chendian	153.0 vs. 57. 9 μg/L, $p < 0.01$	(Peng et al., 2005)
Pb	2010.6	178 children (aged 11–12) (108:70)	Luqiao vs. Chun'an	6.97 vs. 27.8 μg/L, $p < 0.001$	(Wang et al., 2012a)
Pb, Cu, Cd	–	138 persons (59:79)	South China (exposed vs. control)	Pb: 99.83 vs. 92.25 μg/L; Cd: 1.34 vs. 1.32 μg/L; Cu: 0.76 vs. 0.88 μg/L, $p < 0.01$	(Zhang et al., 2007)
Pb	–	187occupational males	Taizhou	100.08 μg/L	(Yang et al., 2013b)
Cr	2004, 2006, 2008	711 chlidren (415:296)	Guiyu vs. Chendian	2004: 120.3 vs. 63.1 μg/L, $p < 0.0001$ 2006: 165.4 vs. 44.1 μg/L, $p < 0.0001$ 2008: 63.4 vs. 28.2 μg/L, $p < 0.0001$	(Xu et al., 2013)
Cr, Mn, Ni	–	144 children (aged 8–13) (71:73)	Guiyu vs. Liangying	Cr: 35.5 vs. 34.1 μg/L; Mn: 20.6 vs. 14.9 μg/L, $p < 0.01$; Ni: 5.3 vs. 3.0 μg/L, $p < 0.01$	(Zheng et al., 2013a)
Hairs					
Heavy metals	2009.12	125 Residents and dismantling workers (86 vs. 39)	Longtang vs. Yuantan	Cd: 1.15 vs. 0.34 vs. 0.05 μg/g; Pb: 40.07 vs. 14.97 vs. 2.94 μg/g; Zn: 138.95 vs.112.51 vs.122.99 μg/g; Cu: 29.81 vs. 17.67 vs. 9.85 μg/g; Ni: 0.74 vs. 0.59 vs. 0.81 μg/g.	(Zheng et al., 2011)
Heavy metal	2007–2008	159 persons (139:10:10)	Taizhou vs. Ningbo vs. Shaoxing	As: 0.423 vs. 0.282 vs. 0.585 mg/g; Cd: 0.940 vs.0.209 vs. 0.223 mg/g; Cr: 1.591 vs. 1.16 vs. 1.15 mg/g; Cu: 53.0 vs.10.7 vs. 10.2 mg/g; Mn: 7.96 vs.1.03 vs. 3.04 mg/g; Ni: 1.77 vs. 0.812 vs. 0.597 mg/g; Pb: 85.3 vs. 2.98	(Wang et al., 2009)

Table 3 (*continued*)

Pollutants	Sampling time	Exposure groups	Sampling sites	Human body burden	References
				vs. 7.60 mg/g. No statistical analysis.	
Urine					
Be, Cu, Mn, Pb, Cd, Zn	2009.11–12	467 workers and residents (20–65 old) (349:118)	Taizhou (exposed vs. control)	Be: 0.02vs.0.01, $p < 0.01$; Cu: 3.00 vs.1.36; Mn: 2.45 vs. 2.77; Pb: 1.80 vs. 1.20; Cd: 1.09 vs. 0.41, $p < 0.01$; Zn: 0.31 vs. 0.34 μg/L.	(Wang et al., 2011a)
Pb, Cu, Cd	–	104 persons (48:56)	Southeast China (exposed vs. control)	Pb: 41 vs. 34; Cu: 26 vs. 25; Cd: 1 vs. 2 μg/g creatinine, $p < 0.05$	(Wang et al., 2011b)
Pb, Cu, Cd	–	138 persons (59:79)	South China (exposed vs. control)	Pb: 34.82 vs. 38.58; Cd: 1.04 vs. 1.98, $p < 0.01$; Cu: 38.06 vs. 23.69 μg/g creatinine	Zhang et al., 2007

could be carried out to prevent inhalation exposure, such as masks, air cleaners, etc. Third, similar to the inhalation situation, in order to prevent soil/dust ingestion and dermal exposure, some safeguard procedures can be also used, especially for the occupational workers (Fang et al., 2013; Xue et al., 2012).

Children and neonates are a particularly sensitive group because of additional routes of exposure (breastfeeding, placental exposures), high-risk behaviors (hand-to-mouth activities in early years, higher risk-taking behaviors in adolescence) and their changing physiology (higher comparative uptakes of air, water and food, and lower toxin elimination rates) (Han et al., 2011; Wang et al., 2012a; Zhang et al., 2011b). The hazardous compounds found in e-waste have strong human body burden effects, especially for neonates and children. The concentrations of heavy metals in placentas and umbilical cord blood in e-waste recycling areas were higher than in the control areas, and some researches showed significant differences between them (Huo et al., 2007; Zheng et al., 2008). It also appears that neonates, due to the mothers' exposure to e-waste, were faced with potential health effects, and e-waste exposure has threatened the neonates in e-waste recycling areas. For example, the neonates from the e-waste exposure areas have been influenced by the heavy metals; these influences include neonate's health, children's health, and changes in cellular expression (Alabi et al., 2012; Li et al., 2011a, 2012, 2008b; Ni et al., 2010).

Consistent correlations between public health effects were hard to assess because of diverse outcomes and variation in exposure variables. For example, as for the blood lead levels in children, even though most results showed that the blood lead levels in children in the exposure areas were significantly higher than those in the control areas (all $p < 0.01$), one study showed no statistical significance. However, we noted consistent associations in studies assessing the effects of e-waste exposure on measures of DNA damage and cellular expression. In all studies, investigators reported higher frequencies of micro nucleated binucleated cells in all the e-waste-exposed populations than in unexposed controls. DNA damage (e.g., DNA injury rates and tail length) was more prevalent in all the e-waste-exposed populations than in controls.

Our ability to assess associations between such heavy metals exposure and the human body burden (or human health) in China was limited by the absence of prospective or longitudinal studies. The associations, by definition, could not be established in cross-sectional studies. Through the literature review, we recorded weak associations between e-waste exposure (heavy metals) and body burden (placenta, umbilical cord blood, blood and serum, hair, urine) or human health (mental health outcomes, children's growth, changes in cellular expression, and DNA effects). Most associations remained significant when we did the comparison between the exposure areas and the control areas. However, the absence of prospective and longitudinal studies, and the small sample sizes (only one study had a sample size >450 people) (Wang et al., 2011b), are concerns. Substantial quantities of pollution are found not only very close to e-waste recycling locations, but, because of contamination of the surrounding environment (specifically air, soil and water) and the resulting food chain pollution, high levels of pollution due to e-waste are also found throughout southern China, including control population locations (Bi et al., 2007; Guo et al., 2010; Han et al., 2011; Zheng et al., 2010).

At the outcome level, when evaluating the concentrations of heavy metals in the human tissues, we for the most part adopted the mean values. Therefore, this could cause some results bias, because of the lack of heavy metals distribution analysis. In addition, several studies that used a retrospective case–control design did not adjust results for confounding variables, especially for the health effects. In one study in particular, investigators did not adjust for the effect of confounders, especially smoking and age, on neonatal outcomes (Guo et al., 2012). Possible alternative influence sources of exposure, including persistent organic pollutants (PBDEs, PCBs, PAHs), pesticides, and industrial pollution, have not been accounted for in the interpretation of results, thereby diminishing the plausibility of certain health outcomes being due exclusively to heavy metals exposure from e-waste. In addition, this study was restricted to China (in fact, to the most relevant researches focused on China), and the researches in other countries (e.g., India, African countries) were not introduced.

Concern about the health influences of heavy metals exposure to e-waste is increasing despite the paucity of solid research. Reported adverse effects include human body burden and human health. Children and developing fetuses are particularly susceptible, and evidence of adverse effects in early life via environmental exposure is increasing. However, few direct-effect studies on the health effects of heavy metals have been undertaken. In addition, noted that the human health of e-waste exposure was not only from the heavy metals, but also from other toxic pollutants of the e-waste, such as PBDEs, PCBs and PCDD/Fs(Li et al., 2007; Ni et al., 2013a; Wang et al., 2012b; Xing et al., 2009). In future, the comprehensive studies of e-waste exposure are suggested to conduct, and the potential effects of other possible factors (smoking and drinking) should be also included.

Due to the global nature of the electronics market and industry, e-waste management and legislative developments in China have significant influence on the environmentally sound management of used electronics at the international level (Chi et al., 2011; Chung, 2012; He et al., 2006; Hu and Cheng, 2013). Under the progressive development of pilot projects and domestic e-waste legislation in China over the past five years, the formal e-waste recycling industry in China has shown considerable growth in both treatment capacity and quality. The growth of the formal sector is important for lessening the environmental and health impacts of e-waste

Table 4
Human health effects of heavy metals from e-waste.

Sampling sites	Exposure groups	Sampling time	Pollutants	Outcomes	References
Neonate's health					
Guiyu vs. Chaonan	2006:152 mother–infant pairs 2007:150 mother–infant pairs	2006 and 2007	Cr, Pb	Birth length:0.50 ± 0.02 vs. 0.50 ± 0.01 m, $p = 0.92$ (2006); 0.50 ± 0.02 vs. 0.50 ± 0.01 m, $p = 0.279$ (2007) Birth weight: 3.12 ± 0.40 vs. 3.29 ± 0.52 kg, $p = 0.05$ (2006); 3.52 ± 0.56 vs. 3.19 ± 0.44 kg, $p = 0.508$ (2007) Apgar scores: 9.77 ± 0.45 vs. 9.79 ± 0.50, $p = 0.82$ (2006); 9.72 ± 0.75 vs. 9.72 ± 0.75, $p = 0.875$ (2007); Behavior: 10.91 ± 0.90 vs. 11.29 ± 0.80, $p = 0.012$; NBNA; 38.46 ± 1.31 vs. 38.92 ± 1.12, $p = 0.043$;	(Li et al., 2008a)
Guiyu vs.Chaonan	220 mother–infant pairs	2008.10–2009.5	Pb, Cd, Cr, and Ni	Birth length:0.494 ± 0.026 vs. 0.497 ± 0.015 m, $p = 0.409$; Birth weight: 3.100 ± 0.488 vs. 3.128 ± 0.441, $p = 0.652$	(Guo et al., 2010)
Guiyu vs. Xiamen	531 mother–infant pairs	2001–2009	Pb	Stillbirth rate: 4.72 vs. 1.03%, $p < 0.01$; Preterm birth: 5.68vs. 5.24%, $p > 0.05$; Birth length: 0.496 ± 0.018 m; Birth weight: 3.168 ± 0.491 vs. 3.258 ± 0.464 kg, $p < 0.01$. Apgar scores: 9.6 ± 0.9 vs. 9.9 ± 0.5, $p < 0.01$;	(Xu et al., 2012)
Children's health					
Luqiao vs. Chun'an	178 children (aged 11–12)	2010.6	Pb	IQ: 109.31 ± 11.81 vs. 110.27 ± 9.97, $p > 0.05$;	(Wang et al., 2012a)
Guiyu vs. Chendian	136 children (aged 3–6)	2006	Pb	IQ: 10.24 ± 2.44 vs.12.92 ± 2.61, $p < 0.01$ (3–4 years old); 13.22 ± 4.52 vs.12.87 ± 3.39, $p > 0.05$ (5–6 years old)	(Han et al., 2007)
Guiyu vs. Liangying	144 school children	–	Cd, Ni, and Mn	FVC: 1859 ± 100 vs. 2121 ± 56 ml, $p = 0.030$ (boys aged 8–9 years); others was not significant.	(Zheng et al., 2013a)
Guiyu vs. Chendian	226 children (<6 years of age)	–	Pb	Height, Weight, Head circumference, Chest circumference ($p > 0.05$)	(Huo et al., 2007)
Guiyu vs. Chendian	303 children (aged 3–7)	2008.1–2	Pb	Activity level: 4.53 ± 0.83 vs. 4.18 ± 0.81, approach-withdrawal: 4.62 ± 0.85 vs.4.31 ± 0.89, and adaptability: 4.96 ± 0.73 vs.4.67 ± 0.83, all $p < 0.01$	(Liu et al., 2011)
Guiyu vs. Chendian	711 children (aged 3–7)	2008	Cr	Height, Weight, Head circumference, Chest circumference ($p > 0.05$)	(Xu et al., 2013)
Adult' health					
Taizhou	187occupational males	–	Pb	FSH: Group (aged ≤30): 5.64; Group (aged 31–45): 11.51; Group (aged 46–60): 15.32 mIU/ml; LH: Group (aged ≤30): 4.59; Group (aged 31–45): 4.90; Group (aged 46–60): 5.96 mIU/ml; T: Group (aged ≤30): 4823; Group (aged 31–45): 4157; Group (aged 46–60): 3562 mIU/ml;	(Yang et al., 2013b)
Changes in cellular expression					
Taizhou	76 (workers and residents)	–	Cd, Pb, Cr, Hg	SOD: 12,401.6 ± 1498.3 vs. 14,076.6 ± 2528.3 U/gHb, $p < 0.05$; MDA: 5.7 ± 1.3 vs. 4.5 ± 0.8 nmol/mgHb, $p < 0.05$; GSH-Px: 57.6 ± 8.5 vs. 71.5 ± 8.5, $p < 0.05$; CD4+: 32.7 ± 6.7 vs.37.8 ± 7.0%, $p < 0.05$	(Zhang et al., 2012)
Guiyu vs. Chaonan	423 mother–infant pairs	2004/2005– 2007	Cd	Placental MT: 67.00 vs. 32.69%, $p < 0.01$	(Li et al., 2011c)
Guiyu vs. Shantou	105 pregnant women	2008.10–2009.6	Cd	Placental MT: 0.058 ± 0.037 vs. 0.038 ± 0.029, $p = 0.003$; S100P: 0.026 ± 0.020 vs. 0.033 ± 0.016, $p = 0.045$; S100 mRNA: 0.175 ± 0.340 vs. 1.462 ± 1.004, $p = 0.000$;	(Zhang et al., 2011a)
Guiyu vs. Chendian	226 children (aged 1–6)	2004.9–11	Pb	Hemoglobin: 127 ± 17 vs. 129 ± 20 g/L, $p > 0.05$	(Xu et al., 2006)
Guiyu vs. Control group	104 (48 workers vs. 56 residents)	–	Pb, Cu, Cd	MNBNCs frequencies: (median: 4.0‰, 1st/3rd quartiles: 2.0–7.0) vs. (median: 1.0‰,1st/3rd quartiles: 0.0–2.0)	(Wang et al., 2011b)

Table 4 (*continued*)

Sampling sites	Exposure groups	Sampling time	Pollutants	Outcomes	References
Guiyu vs. Chaonan	2006:152 mother–infant pairs; 2007:150 mother–infant pairs	2006 and 2007	Cr	DNA damage, $p < 0.05$	(Li et al., 2008a)

treatment in China (Song et al., 2012c, 2013; Zeng et al., 2013). However, due to a range of social and economic factors, some of which have been discussed in this report, informal collectors continue to play a major role in the collection and recycling of e-waste, and informal processing often leads to detrimental effects on both the environment and the health and safety of workers and local communities (Chi et al., 2011; Fujimori and Takigami, 2014; Xu et al., 2012). In the coming years, unfortunately, both formal and informal sectors will probably continue to operate.

5. Conclusions

The widespread production and use of electronic and electrical equipment, the increasing contamination of the environment, and the persistence and bioaccumulation of these heavy metals warrant special consideration of e-waste as an emerging health risk for many populations.

The Chinese government plays a central role in the planning, administration and monitoring of the e-waste recycling system in China. Other actors, including universities and research institutions, individual enterprises, industry associations, NGOs and foreign governments and agencies also play important roles. Effective environmental regulations in e-waste management are needed to prevent excessive exposure to toxicants from e-waste. Improvements in the e-waste management system can thus be achieved through a combination of legislative development and implementation evaluation, technology transfer and innovation, research, knowledge exchange and international cooperation. Both developed and developing countries share joint responsibility in regulating electronic device manufacturing and e-waste transboundary movement. In countries where primitive e-waste recycling processes exist, human health, especially the health of children, needs to drive the regulation and management of recycling activities.

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