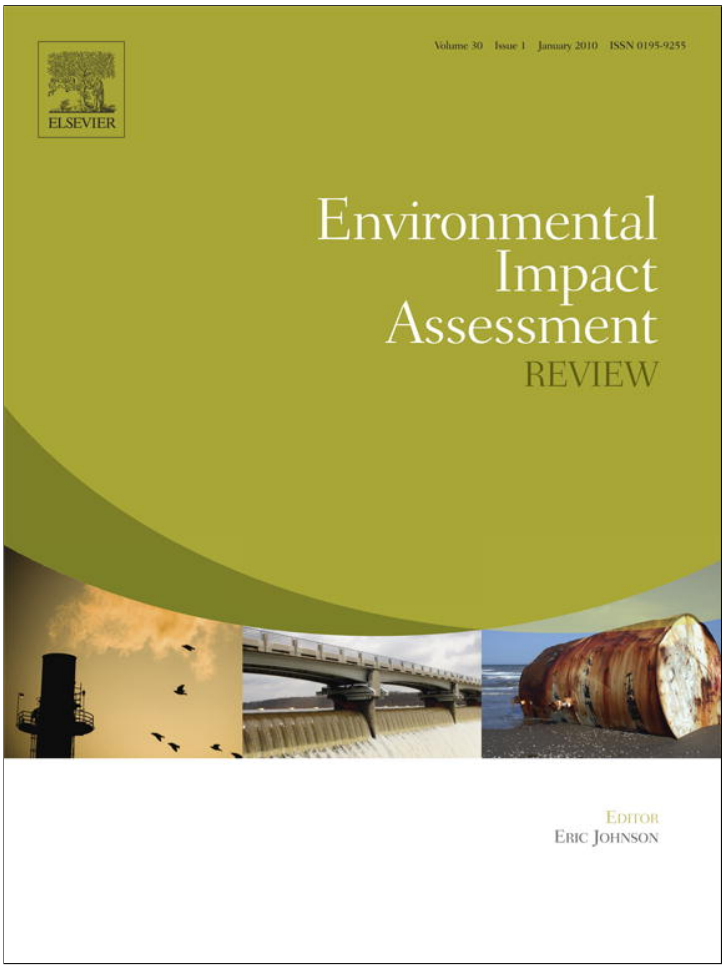

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A review of the environmental fate and effects of hazardous substances released from electrical and electronic equipments during recycling: Examples from China and India

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ABSTRACT

With the increasing global legal and illegal trade of waste electrical and electronic equipment (WEEE) comes an equally increasing concern that poor WEEE recycling techniques, particularly in developing countries, are generating more and more environmental pollution that affects both ecosystems and the people living within or near the main recycling areas. This review presents data found in the scientific and grey literature about concentrations of lead (Pb), polybrominated diphenylethers (PBDEs), polychlorinated dioxins and furans as well as polybrominated dioxins and furans (PCDD/Fs and PBDD/Fs) monitored in various environmental compartments in China and India, two countries where informal WEEE recycling plays an important economic role. The data are compared with known concentration thresholds and other pollution level standards to provide an indication of the seriousness of the pollution levels in the study sites selected and further to indicate the potential negative impact of these pollutants on the ecosystems and humans affected. The review highlights very high levels of Pb, PBDEs, PCDD/Fs and PBDD/Fs in air, bottom ash, dust, soil, water and sediments in WEEE recycling areas of the two countries. The concentration levels found sometimes exceed the reference values for the sites under investigation and pollution observed in other industrial or urban areas by several orders of magnitude. These observations suggest a serious environmental and human health threat, which is backed up by other studies that have examined the impact of concentrations of these compounds in humans and other organisms. The risk to the population treating WEEE and to the surrounding environment increases with the lack of health and safety guidelines and improper recycling techniques such as dumping, dismantling, inappropriate shredding, burning and acid leaching. At a regional scale, the influence of pollutants generated by WEEE recycling sites is important due to the long-distance transport potential of some chemicals. Although the data presented are alarming, the situation could be improved relatively rapidly by the implementation of more benign recycling techniques and the development and enforcement of WEEE-related legislation at the national level, including prevention of unregulated WEEE exports from industrialised countries.

Contents

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1. Introduction

Recent statistics indicate that the total annual global volume of waste electrical and electronic equipment (WEEE) – also referred to as e-waste – is soon expected to reach 40 million metric tones (UNU, 2007). In parallel, there is a dropping lifespan of electronic and electrical products, high consumerism of these products, low recycling rates and illegal transboundary movement from developed to developing countries (Puckett et al., 2002; Brigden et al., 2005; Deutsche Umwelthilfe, 2007; Cobbing, 2008). The number of electronic devices used per capita at the global scale will continue to increase, while their size will further decrease and microprocessors will invade more and more everyday objects (Hilty et al., 2004; Hilty, 2005, 2008). All these facts have triggered an increasing scientific and political interest for how to safely dispose of and recycle WEEE and solutions have been proposed from the perspective of new industrial product designs, manufacturing and recycling philosophies (e.g. the extended producer responsibility, EPR) and green procurement policies. National legislations on WEEE have so far been mainly driven by individual European countries (Sinha-Khetriwal et al., in press) and through the European Directive on WEEE (European Union, 2003a). So far, most developing countries are lagging behind with the development of similar measures (Sinha-Khetriwal et al., 2006) and especially their enforcement.

Restrictions on the use of certain chemicals are included in the EU Directive on Restrictions on Hazardous Substances – RoHS (European Union, 2003b). This Directive has served as a useful guide for other countries, for example China has recently drafted similar administrative measures (National People Congress, 2006). Various multinational collaboration agreements are now effectively in place to ban or limit the movement of certain toxic substances. These include the Stockholm Convention on Persistent Organic Pollutants (POPs) and the Rotterdam Convention on the Prior Informed Consent Procedure for Certain Hazardous Chemicals and Pesticides in International Trade. WEEE also falls under the Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and their Disposal. Despite the existence of these agreements and conventions, the transfer of WEEE from the United States, Canada, Australia, Europe, Japan and Korea to Asian countries such as China, India and Pakistan remains relatively high (Puckett et al., 2002; Terazono et al., 2006; Deutsche Umwelthilfe, 2007; Cobbing, 2008). Moreover, emerging economies such as China and India are themselves large generators of WEEE and have the fastest growing markets for electrical and electronic equipment (Streicher-Porte et al., 2005; Widmer et al., 2005).

WEEE can contain over one thousand different substances, many of which are toxic and some which have a relatively high market value when extracted. Inadequate disposal and poor recycling practices to recover metals such as gold, copper and silver contribute to potential harmful impacts on the environment and pose health risks to exposed individuals. The WEEE stream is thus important not only in terms of quantity but also in terms of its toxicity (Hicks et al., 2005; Widmer et al., 2005). The present review compiles information from published literature about the fate and environmental levels of lead (Pb), polybrominated diphenyl ethers (PBDEs), polychlorinated and polybrominated dioxins and furans (PXDD/Fs) in WEEE recycling areas of China and India, two of the countries most impacted by inappropriate recycling practices and countries that also have a great need for material resources and very low labour costs. Environmental levels of the selected pollutants in the areas of study are compared with some reference toxicological values and the possible impacts for ecosystems and humans in the areas of study are discussed.

2. Emissions from WEEE recycling

WEEE recycling in developing countries is a daisy chain of processes which are carried out in the informal economy. Informal economies can constitute a considerable amount of the gross national product (GNP) of developing or transitional countries (Schneider and Enste, 2003). The activities of WEEE recycling in the informal sector are carried out by a range of legal, unregistered and publicly accepted businesses who give little concern to illegal and clandestinely executed processes which have consequences of great concern to the environment and human health. The businesses collect, sort and manually separate electrical and electronic equipment. The processes involve applying crude methods to segregate substances or material of interest from their original location within the electrical/electronic equipment.

Numerous studies have described various WEEE recycling techniques. These techniques include open burning printed circuit boards (CBs) and cables (Steiner, 2004; Brigden et al., 2005; Gullett et al., 2007; Wong et al., 2007c), burning of CBs for component separation or for solder recovery (Brigden et al., 2005; Wong et al., 2007c), toner sweeping, plastic chipping and melting, burning wires to recover copper, heating and acid leaching of CBs (Hicks et al., 2005; Leung et al., 2006), gold recovery from CBs with cyanide salt leaching or nitric acid and mercury amalgamation (Keller, 2006; Torre et al., 2006; Rochat et al., 2007), and manual dismantling of cathode ray tubes and open burning of plastics (Puckett et al., 2005; Jain and Sareen, 2006). Fig. 1 shows the main toxic substances released during some of these processes and their environmental fate. Three main groups of substances released during recycling can be identified: (i) original substances, which are constituents of electrical and electronic equipment; (ii) auxiliary substances, used in recycling techniques; and (iii) by-products, formed by the transformation of primary constituents. These substances can be found within the following type of emissions or outputs (circles in Fig. 1):

- Leachates from dumping activities
- Particulate matter (coarse and fine particles) from dismantling activities

- Fly and bottom ashes from burning activities
- Fumes from mercury amalgamate "cooking", desoldering, and other burning activities
- Wastewater from dismantling and shredding facilities
- Effluents from cyanide leaching, other leaching activities or mercury amalgamation

Dumped materials containing heavy metals and brominated and chlorinated flame retardants can affect soils (Fig. 1). The mobility of these substances towards other environmental compartments depends on diverse environmental parameters such as pH, organic matter content, temperature, adsorption–desorption processes, complexation, uptake by biota, degradation processes, and the intrinsic chemical characteristics of the substance (Sauvé et al., 2000; Georgopoulos et al., 2001; Hu, 2002; Gouin and Harner, 2003; Qin et al., 2004). Ionic and occasionally, methylated heavy metals, are particularly mobile and bioavailable (Dopp et al., 2004; Hirner, 2006). Lower brominated congeners of flame retardants such as PBDEs are also particularly mobile while higher brominated congeners tend to bond to particles and exhibit lipophilic properties (Gouin and Harner, 2003). PBDEs are used as flame retardants in plastic and textile materials. Three different commercial products exist: PentaBDE, OctaBDE and DecaBDE, which differ in their degree of bromination. All three products can be used in a large variety of polymers, however, PentaBDE has been most widely used in polyurethan foam, OctaBDE in styrene copolymers and DecaBDE in high-impact polystyrene (Alaee et al., 2003). Thus, especially OctaBDE and DecaBDE can be found in WEEE. Heavy metals not recovered during WEEE treatment and residual auxiliary substances like mercury and cyanide can leach through the soil after disposal of effluents and form inorganic and organic complexes within soils (Fig. 1). These effluents can also enter water bodies and the subsequent fate of original and auxiliary substances will depend on the processes described above as well as scavenging processes (between aqueous phase and sediments) and volatilisation.

Dismantling activities release dust particles loaded with heavy metals and flame retardants into the atmosphere. These particles either re-deposit (wet or dry deposition) near the emission source or can be transported over long distances depending on their size. In addition, dust directly incorporated in wastewater can enter the soil or water systems and together with compounds found in wet and dry depositions, can leach into groundwater or react with the biota (Fig. 1). The environmental fate of particles, ashes and fumes containing heavy metals and PBDEs released by burning activities is similar to that of the emissions released by dismantling activities (Fig. 1). However, the thermal or inadequate metallurgical treatment of WEEE can lead to the formation of extremely hazardous by-products such as polyhalogenated dioxins and furans. They are among the most hazardous anthropogenic pollutants (Allsopp et al., 2001; Tohka and Lehto, 2005) and one of their most important formation pathways is the burning of plastic products containing flame retardants and PVC (USEPA, 1997). As copper (Cu) is a catalyst for dioxin formation, Cu electrical wiring coated with chlorine containing PVC plastic contributes to the formation of dioxins (Kobylecki et al., 2001; Gullett et al., 1992). Chlorinated and brominated dioxins and furans (PCDD/Fs and PBDD/Fs), and mixed halogenated compounds like the polybrominated–chlorinated dibenzo-p-dioxins (PBCDDs) and polybrominated–chlorinated dibenzofurans (PBCDFs) can be formed during WEEE burning (Söderström, 2003). Once emitted into the atmosphere, dioxins and furans are dispersed into the environment, and because of their semi-volatile and hydrophobic properties, they tend to accumulate in organic rich media (Adriaens et al., 1995; Smith and Jones, 2000). Higher brominated or chlorinated congeners degrade more slowly and tend to partition more into lipids

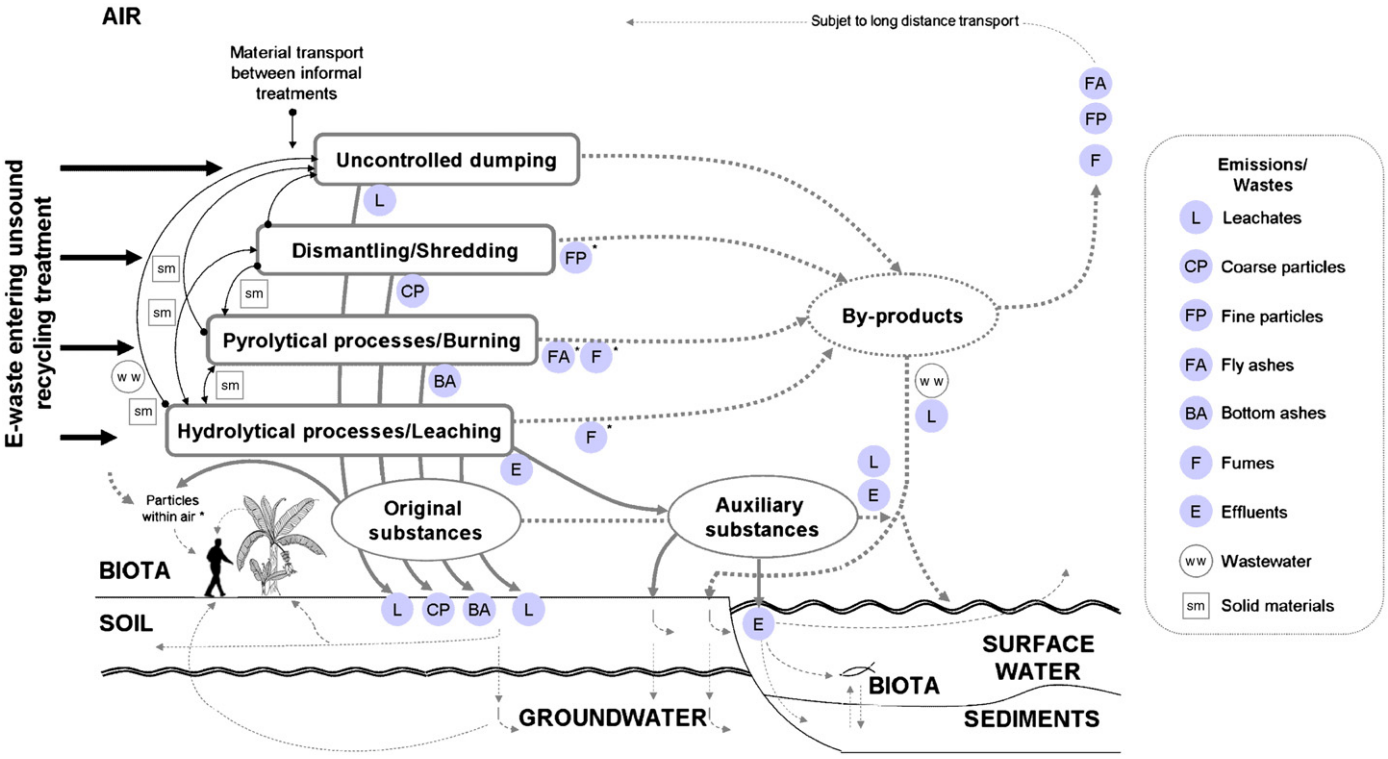

Fig. 1. Principal WEEE recycling activities in China and India, types of produced emissions and general environmental pathways. Ovals: types of substances contained within emissions. Continuous bold lines: fate of *original* and *auxiliary substances*. Dotted bold lines: fate of *by-products* such as dioxins and furans. Black arrows with a bold dot: material transport fluxes between treatments. Fine dashed arrows: general environmental pathways. Environmental fluxes are driven by processes as atmospheric deposition (dry/wet), leaching, adsorption–desorption, complexation (by which heavy metal and cyanide secondary products can be formed), uptake, degradation (chemical/biological) and volatilization. In addition, the environmental fate of pollutants depends on the physico-chemical properties of the media.

Table 1
Literature regarding environmental levels of the selected substances in China and India.

Reference	Analysed substances	Environmental compartments and media monitored	Analytical methods	Location	WEEE recycling operations	Date
Leung et al. (2008)	Heavy metals	Dust (surface dust)	Digestion with HNO_3, ICP–OES	Guiyu, China (recycling workshops, adjacent roads, schoolyard, outdoor food market)	Printed circuit board recycling operations	2004
Huo et al. (2007)	Heavy metals (Pb)	Human blood (children <6 years of age)	GF–AAS	Guiyu and Chendian, China	Dismantling, circuit board baking, acid baths; plastics sorting, including manually stripping	NS
Keller (2006), Rochat et al. (2007)	Heavy metals	Wastewater	NS	Bangalore, India	Cyanide leaching	2006
Puckett et al. (2002)	Heavy metals	Water, sediments, soils	NS	Guiyu, China; Pakistan; India	Acid treatment to recover gold from computer chips, burning and dumping of CBs and wires along the banks of the Lianjiang River, China	2001
Wong et al. (2007a)	Heavy metals	Freshwater	ICP–AES, ICP–MS, Pb isotopic analysis	Guiyu, China (impacted and control zones)	WEEE recycling operations in general (Lianjiang and Nyaniang Rivers), and a strong acid leaching place (Nyaniang River)	2006
Wong et al. (2007b)	Heavy metals	Sediments	ICP–AES, Pb isotopic analysis by ICP–MS. Chemical speciation of Cu, Pb and Zn (mobility and potential bioavailability) by a Tessier sequential chemical extraction	Guiyu, China (impacted and control zones)	WEEEe recycling operations in general (Lianjiang and Nyaniang Rivers)	2005
Wong et al. (2007c)	PBDEs, PCDD/Fs, PAHs, PCBs, heavy metals	Air, soils, sediments	Air PBDEs and PCDD/Fs: USEPA Draft Method 1614 and USEPA Method 1613. Air PAHs, soil PAHs/PCBs, sediment PAHs: GC–MS after Soxhlet extraction; air heavy metals/metalloids, soil and sediment heavy metals: ICP–OES after acid digestion; soil/sediment PBDEs and soil PCDD/Fs: USEPA Method 1614 (draft) and 1613	Guiyu, China	Open burning, acid leaching, reservoir area, rice field, duck ponds, river tributaries, control zones	2004, 2005
Brigden et al. (2005)	Heavy metals, chlorinated benzenes, PCBs, PBDEs, phthalate esters, aliphatic and aromatic hydrocarbons, organosilicon compounds, others	Wastewater, ashes, soils, sediments, dusts	NS	Guiyu, China; New Delhi, India	Manual separation and shredding; removal and collection of solder using heating; acidic extraction of metals; burning of wastes to remove combustible plastics and isolate metals; glass recovery from cathode ray tubes	2005
Deng et al. (2006)	PAHs, heavy metals	Air samples (TSP, $PM_{2.5}$)	Gravimetry, digestion, ICP–OES, AAS	Guiyu, China	Open burning and others	2004
Leung et al. (2006)	PAHs, PCBs, PBDEs, heavy metals	Sediments, soils	Soxhlet extraction, GC–MS, GC–ITMS, microwave digestion, ICP–OES	Guiyu, China	Circuit boards heating, dumping (melted and burnt plastic and discarded printer rollers) on the banks of the Lianjiang River. The authors also sampled in a forested reservoir 6 km away from the WEEE center	2003
Leung et al. (2007)	PBDEs, PCDD/Fs	Soils	Soxhlet extraction, GC–MS	Guiyu, China	WEEE dumping and open burning; acid leaching of CBs	2004
Yuan et al. (2008)	PBDEs and others	Human serum	Extraction with hexane: methyl-*tert*-butyl ether, GC–MS or GC–HRMS according to the bromine number of PBDEs	NS	Mainly dismantling activities	NS
Deng et al. (2007)	PBDEs	Air (TSP and $PM_{2.5}$)	NS	Guiyu, Hong Kong and Guangzhou, China	Heating or opening burning and other activities in Guiyu, and non-WEEE activities in Hong Kong and Guangzhou	2004
Bi et al. (2007)	PBDEs, PCBs, OCPs	Human serum	Gel permeation chromatography (Biobeads S-X3) and GC–MS	Guiyu and Haojiang, China	Chipping and melting plastics, burning coated wire to recover copper, removing electronic components from CBs, burning unsalvageable materials in the open air. The authors also sampled in Haojiang, a nearby area of Guiyu where fishing industry predominates	NS
Luo et al. (2007a)	PBDEs	Fish, sediments	NS	Guiyu, China (Lianjiang and Nanyang rivers)	Open burning, dumping of ashes and wastewater. The authors also sampled in residential areas	NS
Wang et al. (2005)	PBDEs	Soils, sediments	Soxhlet extraction, GC–MS	Guiyu, China	Separation and recovery of metals from circuit boards, PVC-coated wires and cables by open burning	2003
Li et al. (2007a)	PCDD/Fs, PBDD/Fs	Air	Extraction with toluene, HRGC–HRMS	Guiyu, Chendian and Guangzhou, China	WEEE dismantling processes in Guiyu. The other sites does not have a WEEE dismantling industry	2005

(continued on next page)

Table 1 (*continued*)

Reference	Analysed substances	Environmental compartments and media monitored	Analytical methods	Location	WEEE recycling operations	Date
Chan et al. (2007)	PCDD/Fs	Human milk, placenta, and hair from women who gave birth in 2005	Soxhlet extraction (U.S. EPA Method 3540C), HRGC–HRMS. Lipid content in milk/placenta by gravimetry	Taizhou and Lin'an City, China	Open burning, and a control site (Lin'an City)	2005
Luksemburg et al. (2002)	PCDD/Fs	Ashes, sediments, human hair	U.S. EPA Method 1613 (Revision B, dated Sept., 1997)	Guiyu, China	Burning, acid leaching activities. The authors also sampled sediments in areas with a non-direct impact of WEEE recycling	NS

NS: No specified. ICP–OES: inductively coupled plasma–optical emission spectroscopy. GC–MS: gas chromatography-mass spectrometry. GC–HRMS: gas chromatography–high resolution mass spectrometry. HRGC–HRMS: high resolution gas chromatography–high resolution mass spectrometry. GF–AAS: graphite furnace–atomic absorption spectrometry. ICP–AES: inductively coupled plasma–atomic emission spectrometry. ICP–MS: inductively coupled plasma–mass spectrometry. GC–ITMS: gas chromatography–ion trap mass spectrometry.

(Webster and Mackay, 2007). They often deposit near the sources of emission while the lower halogenated compounds are typically transported over longer distances (Fig. 1). In the atmosphere, dioxin and furans are subject to photodegradation and hydroxylation (Watterson, 1999).

This brief description of the environmental fate of specific substances following some recycling methods highlights that inadequate recycling techniques contribute to the pollution of the environment in various ways with potential severe impacts on ecosystems and human health. The extent of the pollution in China and India from these practices is reviewed in the next section.

3. Environmental fate of selected pollutants in China and India

Published literature was reviewed to compile the measured concentrations of lead, PBDEs, dioxins and furans in WEEE recycling sites in India and China. The references, monitored substances, environmental compartments considered, analytical methods used, location of the study, recycling technique used and date of the publication are compiled in Table 1. The following section discusses the concentrations of each of the chemical compounds found from the literature review.

3.1. Lead

3.1.1. Air

Lead (Pb) concentrations reported by Deng et al. (2006) in the air of rural areas of Guiyu, China (TSP and $PM_{2.5}$, total suspended particles with a diameter less than 30–60 μm and particle matter with a diameter <2.5 μm, respectively) exceeded 2.6–2.9 times the upper bracket of air Pb levels for non-urban European sites (<0.15 μg m^{-3}) (World Health Organisation (WHO), 2000) and by 3.1–4.6 times the concentrations of Pb in some metropolitan cities such as Seoul and Tokyo (Fang et al., 2005, Table 2). According to Deng et al. (2006), Pb concentration in Guiyu air was higher than for many other sites in Asia.

Table 2
Lead (Pb) concentrations in total suspended particles (TSP) and particulate matter ($PM_{2.5}$) in air samples of Guiyu, and comparison values.

	Pb (μg m^{-3})
•TSP samples taken on the roof of buildings in WEEE recycling areas in Guiyu, China (Deng et al., 2006) – mean	0.44
•$PM_{2.5}$ samples taken on the roof of buildings in WEEE recycling areas in Guiyu, China (Deng et al., 2006) – mean	0.39
Comparison values	
•Background level for non-urban sites (WHO, 2000)	<0.15
•Urban Asian areas (Fang et al., 2005)	
Tokyo (TSP)	0.125
Seoul ($PM_{2.5}$)	0.096

3.1.2. Bottom ash, dust and soil

The Pb range concentration of 3560–6450 mg kg^{-1} dw in bottom ashes of WEEE recycling facilities in New Delhi reported by Brigden et al. (2005) was 254–461 times higher than the average content of Pb in bottom ash from three major power plants in and around New Delhi (as reported by Sushil and Batra (2006), see Table 3), 7.12–12.9 and 102–184 times higher than the Pb value for industrial soils and the background level for soil (from non-anthropogenic sources) as specified by the State Environmental Protection Administration of China (SEPA, 1995), and ca. 6.72–12.2 times higher than the action value for Pb as stipulated by the Ministry for Social Building, Regional Planning, and Environment Administration of the Netherlands (VROM, 1994) (see Table 3). Exceeding this action value in the Netherlands requires the need for remedial action (Provoost et al., 2006). Lead dust concentrations in CBs from WEEE dismantling and shredding workshops of Guiyu and New Delhi (Brigden et al., 2005; Leung et al., 2008) were higher by factors of 207 to 220 (for Guiyu) and 16.6 to 17.6 (for New Delhi, Table 3) when compared with the Pb action value set by VROM (1994) and the Pb value for industrial soils specified by SEPA (1995). A Pb dust concentration in roads adjacent to WEEE workshops in Guiyu (Leung et al., 2008) also exceeds the Pb

Table 3
Lead (Pb) concentrations in bottom ashes, dust and soils in New Delhi and Guiyu, and comparison values.

	Pb (mg kg^{-1} dw)
Bottom ashes, New Delhi	
•Open burning (wire burning) (Brigden et al., 2005)	3560–6450
Dust, Guiyu	
•From a printer dismantling workshop and from separation and solder recovery workshops (Brigden et al., 2005)	284
	31,300–76,000
•WEEE workers houses (Brigden et al., 2005)	719–4110
•Circuit board recycling workshops (Leung et al., 2008)	110,000
•Adjacent roads to WEEE workshops (Leung et al., 2008)	22,600
Dust, New Delhi	
•WEEE separation workshops (Brigden et al., 2005)	150–8815
•Streets near WEEE recycling facilities (Brigden et al., 2005)	31–1300
Soil, Guiyu	
•Burnt plastic dump site (Leung et al., 2006)	104
Comparison values	
•Bottom coal ash in New Delhi (Sushil and Batra, 2006)	14
•Natural background level for soils (SEPA, 1995)	35
•Level for industrial soils (SEPA, 1995)	500
•Values for soils of Hong Kong (Lau Wong et al., 1993)	75
•Optimum value for soils (VROM, 1994)	85
•Action value for soils (VROM, 1994)	530

action value set by VROM (1994) and the Pb value for industrial soils outlined by SEPA (1995) by 43 to 45 times. The Pb content in dust of streets near WEEE facilities in New Delhi (Brigden et al., 2005) was high when compared to Pb background and industrial levels for soils according to SEPA (1995) and the action value set by VROM (1994) (Table 3). A range Pb dust concentration in WEEE worker's houses in Guiyu (Brigden et al., 2005) was ca. 1.4 to 8.2 times higher than the Pb action level specified by VROM (1994) and the Pb value for industrial soils according to SEPA (1995) (Table 3). Lead soil concentration for WEEE dumping and burning areas of Guiyu (Leung et al., 2006) was higher than the optimum value set by VROM (1994), the Pb background value specified by SEPA (1995) and the value reported by Lau Wong et al. (1993) for soils of Hong Kong (Table 3).

3.1.3. Water

Wastewater containing residues from cyanide and acid leaching processes as well as from WEEE dismantling activities in China (Guiyu, Puckett et al., 2002; Brigden et al., 2005; Wong et al., 2007a) and India (New Delhi and Bangalore, Brigden et al., 2005; Keller, 2006) showed Pb concentrations between 17 and 247 times higher than Pb concentrations reported by Wang et al. (2003) for Pb/Zn ore mining wastewaters in Liaoning Province, China. However some reported values also showed lower concentrations of Pb than in mining wastewaters of Liaoning Province (Table 4). Lead concentration in surface water of the Lianjiang River (Puckett et al., 2002) was found to exceed the concentration for mining wastewaters in China by 10 to 126 times and the drinking water guidelines of WHO (2004a, Table 4) by 190 to 2400 times. Lead in groundwater exceeded the WHO guidelines (2004a, Table 4) by 6.3 times.

3.1.4. Sediments

Sediments collected near discharged residues from WEEE mechanical shredding activities (Brigden et al., 2005) in Lianjiang River showed higher Pb levels than Pb levels found in samples influenced by other WEEE recycling activities like dumping, burning and acid treatments (Puckett et al., 2002; Brigden et al., 2005; Leung et al., 2006; Wong et al., 2007b; Table 5). The shredding-related levels exceeded the Pb mid-range effect guideline for Hong Kong (ISQV-high; Chapman et al., 1999) by 21 to 203 times and the Pb severe effect level within the same guideline (SEL; MacDonald et al., 2000) by 18 to 177 times. Sediments from open burning, dumping and acid leaching areas (Puckett et al., 2002; Leung et al., 2006; Brigden et al., 2005; Wong et al., 2007b) often (but not systematically) exceeded the reference values given in Table 5. On the other hand, sediments from Nanyang River, which is also exposed to WEEE recycling activities, showed Pb concentrations that are lower than the comparison values.

Table 4
Lead (Pb) concentrations in water samples of Guiyu, New Delhi and Bangalore, and comparison values.

	Pb (mg l^{-1})
Guiyu	
•Surface water. Lianjiang and Nanyang rivers – WEEE recycling in general (Wong et al., 2007a)	0.001–0.002
•Surface water: Lianjiang River – area related with circuit board acid and burning processing (Puckett et al., 2002)	1.9–24
•Wastewater from separation of circuit boards and shredding (Brigden et al., 2005)	0.04–46.9
•Wastewater from acid processing (Brigden et al., 2005)	3.20–3.66
•Groundwater in an area of separation of circuit boards and shredding (Brigden et al., 2005)	0.063
New Delhi and Bangalore	
•Wastewater from acid processing (Brigden et al., 2005)	20.4
•Wastewater from cyanide leaching (Keller, 2006)	4
Comparison values	
•Wastewater from mining (Wang et al., 2003)	0.19
•Drinking water guideline (WHO, 2004a)	0.01

Table 5
Pb concentrations in sediments of Guiyu (China), and comparison values.

	Pb (mg kg^{-1} dw)
•Lianjiang River: mechanical shredding (Brigden et al., 2005)	4505–44,300
•Lianjiang River: open burning of circuit boards and wires, dumping and acid operations (Puckett et al., 2002)	300–23,400
•Lianjiang River: acid processing (Brigden et al., 2005)	83–2690
•Lianjiang River: circuit board heating and dumping of WEEE in bank sediments (Leung et al., 2006)	94.3–316
•Lianjiang River (WEEE recycling influence) (Wong et al., 2007b)	230
•Nanyang River (WEEE recycling influence) (Wong et al., 2007b)	47.3
Comparison values	
•Hong Kong ISQV-high (Chapman et al., 1999)	218
•SEL (MacDonald et al., 2000)	250

ISQV: Interim Sediment Quality Value; SEL, severe effect level.

3.2. Polybrominated diphenyl ethers (PBDEs)

PentaBDE and OctaBDEs are complex mixtures of several diphenyl ether congeners. To facilitate comparisons between studies, representative marker congeners for PentaBDE and OctaBDE (i.e. ΣPentaBDE is the sum of BDE-47, -99 and -100 and ΣOctaBDE corresponds to the sum of BDE-183, -196, -197 and -203) were used.

3.2.1. Air

ΣPenta-, ΣOcta-, and DecaBDE values were calculated from Deng et al. (2007) with the aim to be able to compare them with available Predicted Environmental Concentrations developed in the standard risk assessment model "European Union System for the Evaluation of Substances" (EUSES) (ECB, 2001). The monitored values of the most abundant congeners within the air of Guiyu (ΣPentaBDEs) exceeded the corresponding Regional Predicted Environmental Concentration ($PEC_{regional}$), calculated for a densely populated area of 200×200 km with 20 million inhabitants in Europe (Table 6), by factors of ca. 40 ($PM_{2.5}$) and 46 (TSP).

ΣPentaBDEs associated with TSP and $PM_{2.5}$ sampled in Guiyu were approximately two orders of magnitude higher than concentrations monitored in the urban areas of Hong Kong and Guangzhou (places which already have higher levels of these substances than other urban and rural areas around the world, Deng et al., 2007; Table 6) and approximately three orders of magnitude higher than in the air of semi-rural sites in Europe (Lee et al., 2004). In Hong Kong and Guangzhou reported ΣPentaBDEs values were lower than the ECB (2001) $PEC_{regional}$.

Table 6
PBDEs levels in air samples of Guiyu and two urban places of the Pearl River Delta region (China), and comparison values.

	Σ PentaBDE marker congeners[a] (ng m^{-3})	Σ OctaBDE marker congeners[b] (ng m^{-3})	Σ_{ALL}PBDE (ng m^{-3})
TSP (Deng et al., 2007)			
•Guiyu	12.5	0.27	21.5
•Hong Kong	0.09	0.002	0.15
•Guangzhou	0.21	0.004	0.29
$PM_{2.5}$ (Deng et al., 2007)			
•Guiyu	10.9	0.16	16.6
•Hong Kong	0.06	0.001	0.09
•Guangzhou	0.1	0.003	0.16
Comparison values			
•$PEC_{regional}$ for PBDE commercial products (ECB, 2001)	0.27	0.11	

Σ_{ALL}PBDE is the Σ of all analyzed congeners; the $PEC_{regional}$ is a value calculated for a densely populated area of 200×200 km with 20 million inhabitants in Europe (ECB, 2001).

Note: DecaBDE was not measured by Deng et al. (2007).

[a] Sum of BDE-47, -99, and -100.

[b] Sum of BDE-183, -196, -197, and−203.

Table 7
PBDE concentrations in bottom ashes and soils of New Delhi (India), Guiyu and Taizhou (China), and comparison values.

	Σ PentaBDE marker congeners[a] (ng g^{-1} dw)	Σ OctaBDE marker congeners[b] (ng g^{-1} dw)	Σ DecaBDE marker congeners[c] (ng g^{-1} dw)	Σ_{ALL}PBDE (ng g^{-1} dw)
Bottom ashes, New Delhi				
•Burning (Brigden et al., 2005)	NAD	NAD	NAD	23,000
Soils, Guiyu				
•Soils influenced by dumping–burning (Wang et al., 2005; Leung et al., 2006)	21.9	824	NM	1140
•Soils influenced by open burning activities (Wong et al., 2007c)	NAD	NAD	NAD	2906–44,473
•Soils influenced by acid leaching wastewaters (Leung et al., 2007)	244.6	231.9	1270	3570
•Rice crop soils influenced by open burning (Leung et al., 2007)	1.1	3.53	37.3	48.2
•Rice crop soils influenced by open burning (Wong et al., 2007c)	NAD	NAD	NAD	45.1–102
•Reservoir (Leung et al., 2007)	0.475	0.117	2.76	3.8
•Reservoir (Wong et al., 2007c)	NAD	NAD	NAD	2.00–6.22
Soils, Taizhou				
•WEEE recycling site (Cai and Jiang, 2006)	765	12	NM	940
Comparison values				
•Background value for woodland areas of the UK (Hassanin et al., 2004)	NAD	NAD	NAD	12
•Predicted urban value for the UK (Hassanin et al., 2004)	NAD	NAD	NAD	100
•$PEC_{local/regional}$ for PBDE commercial products (ECB, 2001)	7020 (local) 343.2 (regional)	8424 (local) 189.8 (regional)	8476 (local) 70,460 (regional)	NAD

NAD: No available data. NM: Not measured.
Note: Some references did not show ΣALLPBDE concentrations, but instead range references.
[a] Sum of BDE-47, -99, and -100 (some references did not include all three congeners).
[b] Sum of BDE-183, -196, -197, and -203 (some references did not include all three congeners).
[c] BDE-209; Σ_{ALL}PBDE is the Σ of all analyzed congeners; the PEC_{local} represent a worst predicted concentration by modelling in a worst case scenario (such an area of PBDEs production), and the $PEC_{regional}$ is a value calculated for a densely populated area of 200×200 km with 20 million inhabitants in Europe (ECB, 2001).

3.2.2. Bottom ash, dust and soil

Published PBDE concentrations in bottom ash, dust and soils of WEEE recycling areas in New Delhi, Guiyu and Taizhou (Brigden et al., 2005; Wang et al., 2005; Wong et al., 2007c; Leung et al., 2007; Cai and Jiang, 2006) are presented in Table 7 together with some values for comparison. PBDEs identified in dust associated with manual separation of CBs and solder recovery in New Delhi were detected at trace levels (Brigden et al., 2005). Ashes and soils from the New Delhi and Guiyu burning sites had PBDE concentrations that were 230 and 11 to 445 times higher than PBDEs in urban soils of the UK (Hassanin et al., 2004), respectively. Soils in Guiyu affected by acid wastewaters from WEEE leaching techniques also showed a concentration that was 36 times higher than the value for urban soils of the UK. The mean and maximum PBDE concentration in rice crop soils influenced by WEEE open burning activities (Leung et al., 2007; Wong et al., 2007c) was 4 to 8.5 times higher than a reference value for woodland areas in the UK (Hassanin et al., 2004), while soils of a reservoir zone in Guiyu did not exceed any of the reference values used for comparison and reported here.

Soils from Taizhou (a WEEE recycling city in the Zhejiang Province, China) also exceeded the level of PBDEs considered by Hassanin et al. (2004) for urban soils of the UK by a factor of 9 but concentrations were much lower than those found in the soils of Guiyu (Table 7). PentaBDEs were prevalent in soils of Taizhou, while DecaBDEs were prevalent in soils of Guiyu (Table 7). Among the specific commercial product concentrations obtained from the literature, the calculated ΣPentaBDE concentration for Taizhou was 2.2 times higher than a $PEC_{regional}$ of 343.2 ng g^{-1} dw. ΣOctaBDE concentrations for soils of Guiyu were between 1.2 and 4.3 times higher than a $PEC_{regional}$ of 189.8 ng g^{-1} dw (Table 7).

3.2.3. Wastewater

Brigden et al. (2005) reported a wastewater Σ_{ALL}PBDE concentration of 4000 μg l^{-1} from a WEEE shredder workshop that discharged its wastewater via pipes into a shallow channel connected with the

Table 8
PBDEs concentrations in sediments of Guiyu and Hong Kong (China), and comparison values.

	Σ PentaBDE marker congeners[a] (ng g^{-1} dw)	Σ OctaBDE marker congeners[b] (ng g^{-1} dw)	Σ DecaBDE marker congeners[c] (ng g^{-1} dw)	Σ_{ALL}PBDE (ng g^{-1} dw)
Sediments, Guiyu				
•Lianjiang River: wastewater discharged from shredder workshops* and acid processing** (Brigden et al., 2005)	NAD	NAD	NAD	12,000–30,000 6,000–15,000
•Lianjiang River: dumping, acid leaching and burning activities (Wang et al., 2005; Leung et al., 2006)	11.7	3.81	NM	32.3
•Lianjiang River: bottom sediments next to a residential area (Luo et al., 2007a)	74.3	20	30	156
•Nanyang River: bank sediments with burned ashes dumped (Luo et al., 2007a)	6,272	241	35.9	9,357
•Nanyang River: bottom sediments near an open burning site (Luo et al., 2007a)	145.9	6.4	31.1	225
Sediments, Hong Kong				
•Lo Uk Tsuen: bottom sediments receiving wastewater (Luo et al., 2007a)	2.3	2.3	6.0	13.9
•Natural Reserve, Mai Po marshes (Luo et al., 2007a)	<0.3	<0.4	<1	NAD
Comparison values				
•$PEC_{local/regional}$ for the PBDE commercial products (ECB, 2001)	11,700 (local) 84.5 (regional)	20,020 (local) 49.4 (regional)	28,080 (local) 12,844 (regional)	NAD

the PEC_{local} represent a worst predicted concentration by modelling in a worst case scenario (such an area of PBDEs production), and the $PEC_{regional}$ is a value calculated for a densely populated area of 200×200 km with 20 million inhabitants in Europe (ECB, 2001).
NAD: No available data. NM: Not measured.
[a] Sum of BDE-47, -99, and -100.
[b] Sum of BDE-183, -196, -197, and -203.
[c] BDE-209; Σ_{ALL}PBDE is the Σ of all analyzed congeners.

Lianjiang River in Guiyu. This concentration reported by Brigden et al. (2005) is approximately 8 orders of magnitude higher than the range of PBDE concentrations in the dissolved phase of coastal waters of Hong Kong (3.1×10^{-5}–1.2×10^{-4} μg l^{-1}, Wurl et al., 2006) and 7 orders of magnitude higher than Σ_{ALL}PBDE concentrations in water from the Lower South Bay of the San Francisco Estuary (1.03×10^{-4}–5.1×10^{-4} μg l^{-1}), which receives approximately 26% of wastewater treatment plant effluents (Oros et al., 2005). These differences are not surprising as the systems are different (highly concentrated wastewater vs. diluted systems) but the local impact in the Lianjiang River could be considerable, as shown also by the elevated concentrations in sediments concentrations (see below).

3.2.4. Sediments

Table 8 presents Σ_{ALL}PBDE concentrations in sediments influenced by WEEE recycling activities along the Lianjiang and Nanyang rivers in Guiyu, as well as for a place receiving wastewater from a non-WEEE source and for a natural reserve in Hong Kong (Brigden et al., 2005; Wang et al., 2005; Leung et al., 2006; Luo et al., 2007a). Additional information within Table 8 includes calculated concentrations for each commercial product and values for comparison.

The Σ_{ALL}PBDE concentrations presented by Brigden et al. (2005) for sediments influenced by wastewater discharges from WEEE shredder workshops and acid processing (6000–30,000 ng g^{-1} dw) in the Lianjiang River were the highest reported for Guiyu. These concentrations were between 38.5 and 929 times higher than the concentrations presented by Wang et al. (2005), Leung et al. (2006) and Luo et al. (2007a) for sediments of the Lianjiang River, which were impacted by dumping, burning and acid activities, as well as for sediments collected nearby residential zones (32.3–156 ng g^{-1} dw). According to Luo et al. (2007a), bank sediments of the Nanyang River presented higher PBDE concentrations with respect to bottom sediments, while places without the influence of WEEE recycling activities in Hong Kong presented the lowest PBDE levels when compared with data from Guiyu (Table 8). PentaBDE was the most abundant PBDE in sediments from Guiyu. ΣPentaBDEs concentration ranged from 11.7 to 6272 ng g^{-1}, the higher concentrations being some 74 times higher than the corresponding $PEC_{regional}$ (84.5 ng g^{-1}; ECB (2001)). None of the technical products exceeded the PEC_{locals} considered by the ECB (2001) as worst predicted concentrations.

3.3. Dioxins and furans (PCDD/Fs, PBDD/Fs)

3.3.1. Air

Li et al. (2007a) reported PCDD/Fs and PBDD/Fs concentrations in air around Guiyu that ranged from 64.9 to 2365 pg m^{-3} (0.97–51.2 pg of I-TEQ m^{-3}) and from 8.1 to 461 pg m^{-3} (1.6^{-}104 pg of I-TEQ m^{-3}), respectively. According to Li et al. (2007a), these are the highest documented values of these compounds in ambient air in the world and are attributed principally to WEEE dismantling activities. For comparison, PCDD/Fs values reported in other regions range from non detectable to 12 pg of I-TEQ m^{-3} (de Assunção et al., 2005; Lohmann and Jones, 1998; Hassanin et al., 2006), while PBDD/Fs levels documented for Kyoto and Osaka, Japan, range between 1.8 and 12.1 pg m^{-3} and 4.2 and 17 pg m^{-3}, respectively (Hayakawa et al., 2004; Watanabe et al., 1995).

3.3.2. Ashes and soils

Table 9 presents published PCDD/F concentrations in ashes and soils collected in burning and acid leaching sites in Guiyu. The ash component was the most polluted. Maximum total PCDD/F concentration in ashes (Luksemburg et al., 2002) were 13–71 times higher than the total PCDD/F concentration in soils affected by acid leaching activities (Leung et al., 2007) and 14 times higher than the Japanese environmental quality standard for soils established by the Ministry of the Environment (MOE, 2003). PCDD/Fs in soils of a rice crop zone affected by WEEE open burning activities with a daily occurrence and a forested reservoir in Guiyu did not exceed the Japanese standard.

Table 9
PCDD/Fs concentrations in ashes and soils of Guiyu (China) and comparison values.

	Total PCDD/Fs		Total PCDDs and principal congeners concentration		Total PCDFs and principal congeners concentration	
	pg WHO-TEQ g^{-1} dw	pg g^{-1} dw	pg WHO-TEQ g^{-1} dw	pg g^{-1} dw	pg WHO−TEQ g^{-1} dw	Pg g^{-1} dw
Ashes (Luksemburg et al., 2002)						
From burnt and melted plastic	155–14,400	NAD	NAD	NAD	NAD	NAD
Soils (Leung et al., 2007)						
From an acid leaching area	203–1100	12,500–89,800	•Total PCDDs: 16.2 •Principal congener concentration: 10.5 (TCDD)	•Total PCDDs: 3050 •Principal congener concentration: 679 (TCDD)	•Total PCDFs: 489 •Principal congeners concentration: 45.9–281 (TCDF and PeCDF)	•Total PCDFs: 36,250 •Principal congeners concentration: 10,103 – 20,243 (TCDF and PeCDF)
Rice crop soils influenced by open burning	10–13	2320–3130	•Total PCDDs: 3.77 •Principal concengers concentration: 1.51 (TCDD and HxCDD)	•Total PCDDs: 2067 •Principal congeners concentration: 184–625 (TCDD and HxCDD)	•Total PCDFs: 7.96 •Principal congeners concentration: 1.17–4.52 (TCDF, PeCDF and HxCDF)	•Total PCDFs: 667 •Principal congeners concentration: 67.6–396 (TCDF, PeCDF and HxCDF)
Reservoir	0.39–1.5	228–834	•Total PCDDs: 0.14 •Principal congeners concentration: ND–0.059 (TCDD, PeCDD, HxCDD and OCDD)	•Total PCDDs: 429 •Principal congeners concentration: 9.02–390 (TCDD, PeCDD, HxCDD and OCDD)	•Total PCDFs: 0.667 •Principal congeners concentration: 0.041–0.386 (TCDF and PeCDF)	•Total PCDFs: 36.2 •Principal congeners concentration: 11.6–15.6 (TCDF and PeCDF)
Comparison values						
Ecological screening levels (USEPA, 2003a)	NAD	NAD	NAD	0.199	NAD	38.6
Environmental Quality Standard of Japan (MOE, 2003)	1000	NAD	NAD	NAD	NAD	NAD

NAD: No available data; ND: Not determined; PCDDs: Polychlorodibenzo-*p*-dioxins; PCDFs: Polychlorodibenzo-*p*-furans; TCDD: Tetrachlorodibenzodioxin; PeCDD: Pentachlorodibenzodioxin; HxCDD: Hexachlorodibenzodioxin; OCDD: Octachlorodibenzodioxin; TCDF: Tetrachlorodibenzofuran; PeCDF: Pentachlorodibenzofuran; HxCDF: Hexachlorodibenzofuran.

Wong (2006) showed that the total PCDD/F concentration reported by Luksemburg et al. (2002) for open burning sites in Guiyu was 28 times higher than the highest level reported by Minh et al. (2003) for dumping soils in the Philippines.

The United States Environment Protection Agency ecological screening values for dioxins and furans (USEPA, 2003a) were compared with the specific soil concentrations for dioxins (total PCDD) and furans (total PCDF) from WEEE recycling sites in Guiyu, China. Soils influenced by acid leaching activities, rice crop soils and the reservoir area showed total PCDD concentrations that were approximately 15,300, 10,400 and 2160 times higher than the ecological screening level for total PCDDs, while total PCDF concentrations in soils affected by acid leaching and in rice crop soils were 939 and 17 times higher than the corresponding ecological screening value for furans (Table 9).

The PCDD/F homologue profiles in soils of Guiyu were dominated by TCDDs, TCDFs and PeCDFs in soils affected by acid leaching, and by TCDDs, OCDDs, TCDFs and HxCDFs in the rice crop and reservoir soils. PCDF concentrations were higher than PCDD concentrations (Table 9).

3.3.3. Sediments

Luksemburg et al. (2002) reported total PCDD/F concentrations in Lianjiang's riverbank sediments which were influenced by WEEE recycling activities in Guiyu, near residential areas, and downstream zones (20–50 km away from recycling sites). The observed concentration patterns were that riverbanks with dumped ash had concentrations (35,200 pg WHO-TEQ g^{-1} dw) greater than concentrations in sediments in residential areas near the dumped ash (21.2–2690 pg WHO-TEQ g^{-1} dw) which in turn had concentrations greater than sediments in dowstream areas (1.69–3.49 pg WHO-TEQ g^{-1} dw). The total PCDD/F concentrations reported by Luksemburg et al. (2002) were 7 to 2514 times higher than sediment PCDD and PCDF values in Suzhou Creek (2.9 to 14 pg WHO-TEQ g^{-1} dw), a major natural waterway that passes through Shanghai (Li et al., 2007b). Moreover, the value for sediments with dumped ash was 291 times higher than concentrations for sediments collected in the Elbe River near the Spolana chemicals factory and sewage treatment works (121–140 pg WHO-TEQ g^{-1} dw) after the Elbe flood event of 2002 (Stachel et al., 2004) in Europe.

4. Environmental and health perspectives in China and India related with WEEE recycling activities

Until recently, it has been difficult to clearly link environmental pollution with WEEE recycling activities. However, as summarized in this review, many studies published over the past few years clearly indicate a causal relation between pollution levels and emissions from informal WEEE recycling activities. Atmospheric pollution due to burning and dismantling activities seems to be the main cause for occupational and secondary exposure at WEEE recycling sites. Generally speaking, a growing body of epidemiological and clinical evidence has led to an increased concern about the potential damaging effects of ambient air pollution on health (Brook et al., 2004). Combustion typically generates smaller particles (PM<2.5 μm in diameter) (Cormier et al., 2006) and consequently, fine particulate matter ($PM_{2.5}$, strongly implicated in pulmonary and cardiovascular disease) within Guiyu exceed the USEPA 24-h $PM_{2.5}$ ambient air quality standard, the $PM_{2.5}$ summer mass concentrations in Shanghai (Ye et al., 2003; Qiu et al., 2004; National Ambient Air Quality Standards, 2006; Deng et al., 2007) and present higher levels of Pb, PBDEs, PCDD/Fs and PBDD/Fs than coarser particles (TSP). Among the direct and indirect exposed groups to $PM_{2.5}$, the more vulnerable are pregnant women and children. Eighty percent of children in Guiyu suffer from respiratory diseases and they are particularly vulnerable to Pb poisoning (Baghurst et al., 1992; Wasserman et al., 1998; Guilarte et al., 2003; Grigg, 2004; Needleman, 2004; Qiu et al., 2004; Jain and Hu, 2006). Blood lead levels (BLLs) in children of Guiyu (15.3 μg dL^{-1}) exceed the Chinese mean (9.29 μg dL^{-1}) thus posing a potentially serious threat to children's health; air pollution probably being the cause for this (Wang and Zhang, 2006; Huo et al., 2007).

Residents of Guiyu are also exposed to PBDEs (the highest BDE-209 concentration in serum of electronics dismantling workers of Guiyu is the highest ever reported in humans; Bi et al., 2007) and dioxins (total PCDD/F intake doses in Guiyu far exceed the WHO 1998 tolerable daily intake limit and daily intake limits in areas located near medical solid waste incinerators; Nouwen et al., 2001; Domingo et al., 2002; Li et al., 2007a), and again children and child-bearing women are particularly vulnerable (daily dioxin intake doses of children in Guiyu are about 2 times that of adults, and an elevated body burden in child-bearing women of Taizhou may have health implications for the next generation; Chan et al., 2007; Li et al., 2007a). According to Yuan et al. (2008), the median concentration of total PBDEs in serum of WEEE dismantling workers of Guiyu was twice as high than that of a control group (from a village located 50 km away of Guiyu). Although studies like the one of the Hong Kong Environmental Protection Department (HKEPD, 2000) showed that less than 2% of human dioxin intake is from direct inhalation, a study of Chan et al. (2007) suggests that people from the WEEE recycling site of Taizhou, China, are more exposed to the toxic chemicals via inhalation, in addition to dermal contact and consumption of local foods. This is due to the relatively high background contamination levels in the air.

Human exposure to dioxins begins with atmospheric emissions (Beck et al., 1994), of which incineration releases the largest quantity (WHO, 2004b). Dioxin levels in hair reflect those in the atmosphere (Schramm et al., 1992; Tirler et al., 2001; Nakao et al., 2002; Nakao et al., 2005). Luksemburg et al. (2002) reported total PCDD/F concentrations in hair samples of people living near WEEE recycling facilities in Guiyu that ranged between 16.4 and 25.6 pg WHO-TEQ g^{-1} dw and were similar to the lower PCDD/F value reported for hair samples from a very contaminated pentachlorophenol site in China (12 and 120 pg WHO-TEQ g^{-1} dw; Luksemburg et al., 1997) and about 29 to 466 times higher than the PCDD/F level of people exposed to ambient air in Tsukuba and Ryugasaki, Japan (0.56 pg WHO-TEQ g^{-1} dw; Miyabara et al., 2005). Besides the direct impact of dioxins and furans on the human population and the environment of Guiyu, there is evidence of transport of PCDD/Fs and PBDD/Fs from the WEEE recycling site of Guiyu to the nearby area of Chendian (Li et al., 2007a).

Unlike fine particulate matter, larger coarse dust particles (from 2.5 to 10 micrometers in diameter) do not usually reach the lungs of humans, but they can irritate the eyes, nose and throat (USEPA, 2003b). Furthermore, the metal bioavailability factor (like Pb) for dusts is higher than other environmental sources of exposure like soils (Rasmussen, 2004). The transport of metallic dust and dust containing PBDEs into areas outside the WEEE recycling site such as nearby streets or WEEE recycling workers' houses in New Delhi and Guiyu suggest there is also a risk of secondary chemical exposure. In an investigation by Leung et al. (2008) into the presence of seven heavy metals in dust of printed circuit boards of recycling workshops in Guiyu, levels of Pb, Cu, and Zn were found to be very high. These authors also sampled dust at a schoolyard and an open air food market within Guiyu. They reported elevated concentrations at these places including Pb and Cu levels which exceeded the Canadian residential/park guidelines for Pb and Cu (EC, 1999) by 3.3–6 and 2.5–13.2 times in the case of the schoolyard, and Cu, Ni, Pb, and Zn which exceeded the New Dutch List optimum values (VROM, 2001) for these metals by 10, 5.4, 16, and 4.5 times respectively, in the case of the open air food market. Overall Leung et al. (2008) found that the hazard quotient for Pb was highest at their studied locations (contributing to 89–99% of the risk). High heavy metal values at the open air food market are a concern because food market items (i.e., vegetables) which are often placed on top of newspapers or in plastic buckets on the ground could easily come into contact with contaminated dust especially during the

dry season (Leung et al., 2008). Moreover, in comparison to adults, the potential health risk for children is eight times greater, and since children sometimes accompany their parents to the workshops, they can become even more easily exposed to metal-laden dust (Leung et al., 2008). Other research issues within a risk assessment framework should be investigated, including bioaccessibility of heavy metals in dust (mobilization of contaminants from ingested dust) and oral bioavailability of heavy metals in dust (contaminant fraction that reaches the systemic circulation) (Leung et al., 2008).

Ashes are another hazardous emission of burning activities and are considered as a further potential risk factor for environmental and human health in the WEEE recycling locations reviewed in this paper. According to Lundin (2007), the highest concentrations of dioxins are found in ashes, and among these, fly ashes contain much higher concentrations than bottom ashes (Petrlík and Khwaja, 2006; Lundin, 2007). The literature reviewed for the case studies in China and India did not evaluate heavy metal and persistent organic pollutants concentrations within fly ashes. However the literature did report levels for bottom ashes which far exceed values for ash from major power plants, soil action values, values for industrial (Pb) and urban soils (PBDEs), as well as environmental quality standards and ecological screening values (PCDD/Fs). Even though a number of studies have shown a low leaching capacity of toxic substances from bottom ash, it must be considered that leaching potential tests are, in most cases, carried out in ideal laboratory conditions and do not necessarily correspond to the fate of wastes in the environment where they are deposited (IPEP, 2006). It has been proven, for example, that the leaching potential of PCDD/F increases with increasing dissolved humic matter and pH (Kim et al., 2002). Potential impacts of toxic ashes can also include the capacity of some heavy metals and additive PBDEs to leach out of the ash and contaminate other environmental compartments (Rahman et al., 2001; Rai et al., 2004).

Other emissions from WEEE recycling, such as leachates and toxic liquids, increase human risk of exposure through impacted natural resources such as soils, crops, drinking water, livestock, fish and shellfish. Soil contamination is particularly important in Guiyu, where rice is still cultivated despite the town's conversion to a booming WEEE recycling village since 1995 (Azuma, 2003). About 65% of Pb, Cd and Cr are likely to accumulate in the edible part of rice, the endosperm (Dong et al., 2001). High concentrations of PBDEs in soils of rice fields of Guiyu indicate that, as these compounds are persistent in soils and vegetation, slow uptake may be occurring over extended timescales, so that levels in biota may increase with time (ECB, 2001; Gouin and Harner, 2003). Total PCDD concentrations reported for soils of acid leaching sites, rice crops and a forested reservoir in Guiyu far exceed ecological screening levels (USEPA, 2003a). The homologue dioxin and furan profiles in soils of Guiyu were dominated by TCDDs, TCDFs, PeCDFs, HxCDFs and OCDDs. Among these kinds of dioxins and furans, the TCDDs and TCDFs pose the highest toxicity (Söderström, 2003; Schecter et al., 2006). As the consumption of food is one of the most important sources of human exposure to PBDEs, PCDD/Fs and PBDD/Fs (contributing more than 90% of total exposure in the case of dioxins and furans with fish and other animal products accounting for approximately 80%), bioaccumulation of these substances in red meat, milk, eggs, fish and shellfish must be considered as a matter of high concern in the places studied (Commoner et al., 2000; Bocio et al., 2003; Birnbaum and Staskal, 2004; Petrlík and Ryder, 2005). Chan et al. (2007) found that consumption of foods of animal origin (especially crab meat and eggs) is the main dietary exposure to dioxins at a WEEE recycling site in China (Taizhou). According to Luo et al. (2007a,b), PBDE concentrations in fish and shellfish in the Nanyang and Lianjiang rivers were 10–15,000 times higher than levels reported for other regions (the lower BDE-47 and -28 being the most abundant congeners in carps and tilapia) and about 200–600 times higher than PBDE levels in bottom sediments collected in the same rivers.

Wastewater containing dismantling and shredding residues and other toxic liquids from WEEE recycling activities (such as acid and cyanide leaching) represent a serious threat to ecosystems and human health. According to Wong et al. (2007a), the riverine environment of Guiyu is heavily impacted by WEEE-related activities. Temporal distributions of dissolved heavy metals suggested recent discharges of metals attributable to a strong acid leaching operation of WEEE along the Lianjiang and particularly Nanyang rivers within Guiyu, where dissolved Ag, Cd, Cu and Ni were significantly elevated. Pb isotopic studies also confirm that non-indigenous Pb is present in the Lianjiang and Nanyang rivers (Wong et al., 2007a). Even though another contributor to the increase in dissolved metals within these rivers can be the discharge of untreated domestic wastewater, it is suspected that it is only partially responsible for water quality degradation and may represent trivial importance in terms of dissolved metal concentrations in the riverine systems (Wong et al., 2007a). Dissolved metals are considered to be the most mobile, thus reactive and bioavailable fractions in an aquatic system and are cause for concern (Wong et al., 2007a). The fact that the rivers above are still used for agriculture and aquaculture represents a major health threat to the local community (Wong et al., 2007a). Groundwater in Guiyu presents high Pb levels when compared to the WHO guidelines cited above. In fact, due to the level of local drinking water pollution, water is being trucked in from the town of Ninjing, 30 km away from Guiyu (Westervelt and Puckett, 2003). Concerning sediments, a study of Wong et al. (2007b) showed that the sediments of the Lianjiang River contribute significantly as a source of non-indigenous Pb. Lianjiang River sediments have higher sediment metal concentrations than the sediments of the Nanyang River. This could be due to the fact that the Nanyang River has a lower pH than the Lianjiang River and that dumping of strong acids into the Nanyang River could have lowered its pH and thus increased metal solubility, hence reducing metal absorption and increasing bioavailability (Wong et al., 2007b). Sediments affected by wastewater discharges from WEEE shredder workshops (with high concentrations of Σ_{ALL}PBDEs) and acid processing in Guiyu also showed high PBDE concentrations. As wastewater is discharged into the Lianjiang River and into channels connected with it, further monitoring is warranted for this river to determine precisely the extent of pollution to aquatic organisms and implications for drinking water purposes or for recreational purposes. The rivers studied are part of the irrigation network from which water is extracted for crop irrigation (Wong et al., 2007b).

Given the above, some active measures of environmental and occupational protection should be put in place by introducing advanced processing methods, improving the workplace environment, and biomonitoring of the exposed populations (Yuan et al., 2008).

5. Policy considerations

The complexity of composition of electrical and electronic equipment imposes significant and new challenges for recycling. The complex connections between substances are often difficult to break up and separate due to limitations in separation physics as well as incompatible thermodynamics. It also means that often conflicting technical interests have to be solved: recovering certain substances can lead to the inevitable loss of others (Reuter and Verhoef, 2004; Hagelüken, 2006).

Complex compositions, huge logistical challenges, and an often suboptimal organisation of the sequence of recycling stages can render the complete recycling chain uneconomical, subject to product type. High environmental and social standards required for recycling operations in e.g. the USA, Japan and the EU increasingly trigger illegal exports of WEEE from industrialised and post-industrialised countries to developing and transition countries. There they are either partly reused, dumped immediately or processed in the above described

"backyard" recycling operations in an uncontrolled environment. The "economic driver" for these illegal or doubtful exports has various facets: taxes are usually circumvented and labour costs are only a fraction of those in industrial countries. These factors in combination with low standards or the absence of standards for environmental impact and protection as well as health and safety, leads to much cheaper WEEE treatment costs in backyard recycling facilities compared to state-of-the-art industrial plants. For the latter, investment and operational costs for environmentally sound treatment make up a significant share of the treatment costs. On the other hand, their use of sophisticated, large-scale processes enables the recovery of valuable substances such as precious metals with a much higher yield than backyard operations, which for relevant parts such as circuit boards usually overcompensates the cost disadvantages (Rochat et al., 2007). Tipping the scales often requires a mixed calculation between reuse value and backyard recycling. If, for example, at least a certain portion of devices or contained components from a container of scrapped computers can be sold in the importing country for reuse, the sale revenue generated might statistically overcompensate for all the inefficiencies in the system. The bulk part which cannot be reused is then sold in the importing countries to backyard operators (mainly in Asia) or simply incinerated and dumped (mainly in Africa). The main profit out of this is kept in the hands of unscrupulous traders on both sides of the ocean, with the informal sector usually obtaining only a small portion of the value-added in the whole chain while bearing all the health and safety risks.

Furthermore, even in the industrialised and post-industrialised countries the large majority of small EEE devices end up in the waste bin (UNU, 2007). All these hidden WEEE streams lead to significant, irrecoverable losses of valuable scarce resources and lead to significant environmental damage.

Though the recycling of WEEE is already anticipated as an increasing problem of transnational and partly global dimensions, only a minority of the world-wide population is covered by regional or even local WEEE policy measures. Most of these policy incentives such as the EU's WEEE Directive are dominated by looking at ways to: 'do good for the environment' with the EPR (Extended Producer Responsibility) principle as a starter. At the time of the development of the Directive in the mid 1990s, the focus was primarily on control over toxic substances by means of smart Design for Recycling (DfR) and manual disassembly of hazardous components in the recycling phase itself. As a result, the WEEE Directive prime environmental strategies have become:

- Weight based recycling targets
- A single collection amount of 4 kg per inhabitant
- An origin-oriented categorization of products (Annex I)
- Selective treatment rules (by manual dismantling) for recyclers (Annex II)

However, more than 10 years later, experiences show that WEEE policies should serve multiple and broader environmental goals. Significant developments in shredding and separation technologies suggest that dismantling as such, does not bring the desired toxic control as it depends much more on the destinations of disassembled components and/or shredder fractions, plus there are relatively high costs involved. In addition, technological progress in dedicated smelting and refining operations have resulted in improved yields for a wide range of metals while simultaneously safely preventing emissions of hazardous substances (Hagelueken, 2006). Increasing focus is now placed on optimizing interfaces between dismantling, shredding/sorting and integrated metals smelting. In this context, the recovery of valuable materials (prevention of new material extraction also decreased emissions) and energy preservation have become much more important. At last, a more practical categorization of material streams with similar content in (precious) metal, glass and plastic dominated products occurred naturally, instead of a division by 'origin' as in Annex I of the WEEE Directive.

In India, no specific law regulates WEEE recycling yet, but the Indian government is drafting a WEEE legislation. A lack of control and regulation of the WEEE recycling industry has led the poorest strata of the population to find an economic benefit in recovering the valuable parts of WEEE with unprofessional methods while simply dumping the non profitable and often hazardous components of WEEE products. Though this sector makes its living out of these hazardous processes, it is paramount to promote the integration of the informal sector in the WEEE management system and to increase eco-efficiency by implementing appropriate recycling procedures (Widmer et al., 2005). This includes the creation of transparent and in-praxis workable interfaces between "informal" collection and dismantling/pre-processing with industrial-scale metals recovery and toxics control from complex critical parts (such as circuit boards). This is also elementary for not further losing substances used in EEE due to inappropriate recycling procedures (Rochat et al., 2007).

Another issue of concern in India is that the government trade statistics do not distinguish among imports of new and old computers as well as peripheral parts. For this reason it is difficult to track what share of imports is used. Furthermore, domestic WEEE is significant and will contribute a growing amount to the overall WEEE in India in addition to the continuing illegal imports. There are only three licensed hazardous waste dumps in the entire country, and much solid waste containing heavy metals and other hazardous substances is landfilled (Bortner, 2004). India has started to work on its national WEEE Management and Handling Rules, but a clear roadmap has not yet been communicated.

China has historically been one of the largest recipients of WEEE. However, recent initiatives by the Chinese government have reduced imports (Bortner, 2004). Other regulations and action plans concerning WEEE in China have been drafted, but deficiencies are obvious. Extended producer responsibilities (EPR) have been introduced but are not well defined. Eight formal facilities have been planned and are under construction or in operation along the eastern coast of the country, but it will be difficult for them to compete with the informal processes (Liu et al., 2006). WEEE recycling and disposal is typically disorganized at present and the legislation to regulate it has not yet been finalized. Currently, the majority of WEEE in China is processed in backyards or small workshops using primary methods such as manual disassembly and open burning. Unlicensed processes are mainly located in the southern Guangdong province and in Zhejiang province in eastern China (Liu et al., 2006).

China proposed Regulations on the Recycling and Treatment of Waste Household Electrical and Electronic Appliances, which were originally intended to come into effect on 1st May 2008, but which are likely to be delayed. In addition, China is expected to publish a catalogue listing substances which are subject of restrictions and compulsory certification. This is part of the Chinese "Measures for Administration of the Pollution Control of Electronic Information Products", the so-called Chinese RoHS. The methods of the Chinese RoHS shall apply control and reduction of pollution and other public hazards to the environment caused during the production, sale, and import of information technology products in the People's Republic of China. However, these methods shall not apply to the manufacturing of products destined for export. The Chinese RoHS regulations do not apply to Hong Kong or Macao, only to mainland China. This should be viewed critically, because Hong Kong has already become the hub of used EEE shipments.

Though both countries, India and China, are developing countermeasures against WEEE imports and for environmentally sound handling of WEEE, these measures require additional efforts at the local, regional, national and international levels. This requirement mainly results from the rather complex transnational supply chain of EEE. In this context, improved downstream monitoring of European

and North-American WEEE up to the final destination and the prevention of illegal or doubtful exports can substantially contribute to lessen hazardous emissions from global WEEE, as well as to improve the recovery of valuable substances contained therein.

6. Conclusion

This review of data on the environmental fate and effects of hazardous substances released from WEEE during informal recycling operations in China and India suggests a causal relationship between the release of Pb, PBDEs and dioxins/furans and the determined concentrations in environmental components (e.g. soil and air), biota and humans. The comparison with reference values from various national and international standard documents leads us to the assumption that emissions originated from these recycling operations cause serious detrimental effects on humans and to the environment. Most affected are WEEE recycling workers through direct exposure to Pb, PBDEs and dioxin pollution in the ambient air. However long-range transport of pollutants was observed as well, which suggest a risk of secondary exposure also for remote areas. Leachates from bottom ashes, informal dump sites and toxic liquids from acid and cyanide leaching activities have been identified as the other important source for the contamination of environmental compartments and an increased human exposure through affected natural resources such as soils, crops, drinking water, livestock, fish and shellfish.

These findings clearly indicate an urgent need for better monitoring and control of the informal recycling sector in China and India. However, since the livelihoods of large population groups depends on the income from recycling activities, it is paramount to include the informal sector into formal WEEE recycling systems instead of trying to eliminate the informal sector. Possible solutions should include the creation of transparent and in-praxis workable interfaces between "informal" collection and dismantling/pre-processing with industrial-scale material recovery and control of hazardous fractions.

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