Physical processing, dismantling and hydrometallurgical treatment of e-waste: A systematic review of risks to occupational and public health

Ed Cook[a], Costas A. Velis[a,]*, Spyridoula Gerassimidou[a], Aditi Ramola[b], Arne Ragossnig[b]

[a]*School of Civil Engineering, University of Leeds, Leeds, LS2 9JT, UK*

[b]*International Solid Waste Association, Stationsplein 45 A4.004, 3013AK, Rotterdam, Netherlands*

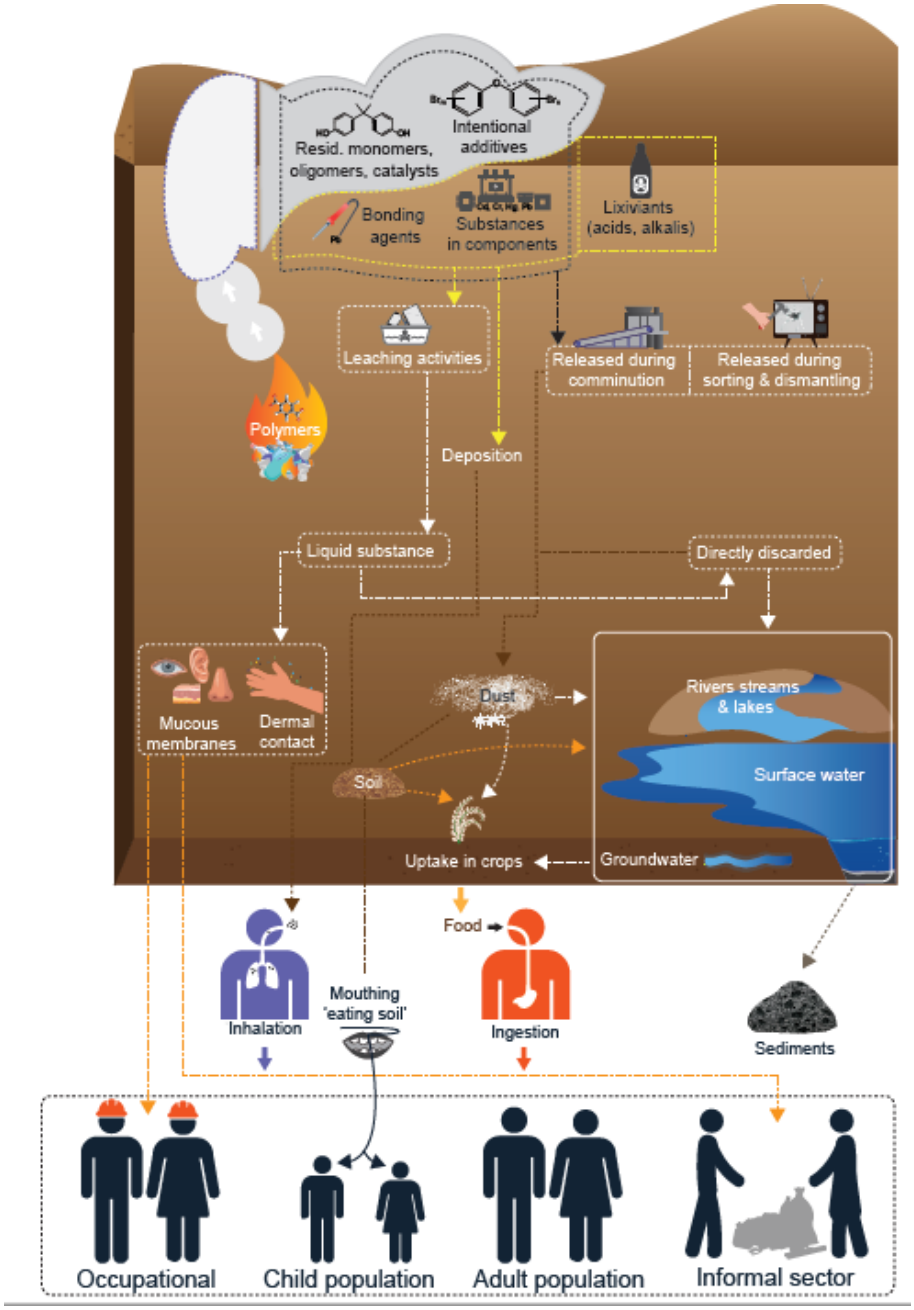
*Corresponding author: c.velis@leeds.ac.uk; Telephone: +44 (0) 113 3432327; Room 304, School of Civil Engineering, University of Leeds, Leeds, LS2 9JT, United Kingdom

Abstract

Across the Global South, electrical and electronic waste (e-waste) is recovered using rudimentary and often dangerous methods in informal and unregulated facilities. Although these activities provide a valuable contribution to the global circular economy, their uncontrolled nature results in a risk of potentially hazardous substance emission into the environment from where they may pose considerable risk to both occupational and public health. Here, we focus a systematic PRISMA review on two distinct groups of activities undertaken in e-waste management in low- and middle-income countries (LIMICs): (i) Physical deconstruction and reclamation, involving dismantling assemblies of items and materials to recover value; and, (ii) hydrometallurgical treatment, involving the dissolution and suspension of precious metals using solvents (cyanide) and acids (aqua regia). For comparison purposes, we consolidate information on (i) and (ii) according to the types of substances evidenced; and identify, critically assess and rank most prevalent hazard-pathway-receptor (H-P-R) risk combinations experienced by people working across the Global South. Despite the proliferation of publications, evidence to assess risk is comparatively limited. Still, we are confident to highlight the extremely hazardous nature of work undertaken, often by children, handling highly hazardous substances without protective equipment to reclaim gold and other precious metals using hydrometallurgical processes. Emissions of hazardous substances, particularly potentially toxic elements (PTEs) from physical dismantling also represent a serious risk to health. Numerous sources speculatively link concentrations in the environment (a significant risk to children who have a tendency to eat soil) to e-waste dismantling processed. However, many of the sources that identify elevated substance concentrations in environmental media face difficulties in unambiguously and convincingly linking emissions from specific activities to the environmental concentrations, i.e. establishing causality. This key limitation presents us with a challenge for designing and implementing interventions to target, control and replace such highly risky resource recovery methods. Yet, such insufficient information cannot be used as an excuse for inaction, especially as our generalised H-P-R inferences here provide for sufficient interlinkages.

Keywords: Health and safety; E-waste; WEEE; Waste; Informal recycling sector; Recycling; Resource recovery; Circular economy; Global South; Risk; Hazardous waste; Hydrometallurgical processing; Potentially toxic elements; Low- and middle-income countries; Poverty alleviation; SDGs; Systematic review.

Graphical abstract



Abbreviations

ADI	'acceptable daily intake'
As	arsenic
BDE	brominated phenyl ether
BFR	brominated flame retardants
Cd	cadmium
Co	cobalt
CP	chlorinated paraffins
CR	Cancer risk
Cr	chromium
CRT	cathode ray tube
Cu	copper
dis.	dismantling
DP	Dechlorane Plus
DRC	dioxins and related compounds
EU	European Union
Fe	iron
Geog.	geographical context
Haz.	hazard
Hg	mercury
HIC	high income countries
HQ	hazard quotient
HS	Harmonized Commodity Description and Coding System
HSE	Health and Safety Executive
IARC	International Agency for Research on Cancer
ICT	Information and Communication Technologies
In	Indium
I-TEQ	international toxic equivalency factor
L	likelihood
LCCPs	long chain paraffins
LIC	low income countries
LIMIC	low income and middle income countries
LMC	lower middle income countries
MCCP	medium chain paraffins
Mn	manganese
Mo	Molybdenum
MSW	municipal solid waste
n	number of samples
Ni	nickel
OECD	Organisation for Economic Co-operation and Development
PAH	polycyclic aromatic hydrocarbons
Pb	lead
PBDEs	polybrominated diphenyl ethers
PCB	polychlorinated biphenyls
PCDD	polychlorinated dibenzo-p-dioxins
PCDD/Fs	polychlorinated dibenzo-p-dioxins and polychlorinated dibenzo-p-furans
PCDF	polychlorinated dibenzofurans
PM	particulate matter
PM_{10}	particulate matter < 10 µm
$PM_{2.5}$	particulate matter < 2.5 µm
POPs	persistent organic pollutants
PPE	personal protective equipment
PPP	purchasing power parity
PRISMA	preferred reporting items for systematic reviews and meta-analyses

PTEs	potentially toxic elements
PVC	polyvinyl chloride
R	risk
rec.	recycling
Ref.	reference
RQ	research question
S	severity
Sb	antimony
SCCP	short chain paraffins
TBBPA	tetrabromobisphenol A
TEQ	toxic equivalency
Tl	thallium
UMC	upper middle income countries
US	United States
USD	United States dollars
WEEE	waste electrical and electronic equipment
wt.	Weight (i.e. on a weight reporting basis)
Zn	zinc

1. Introduction

Electronic and electrical waste (hereafter e-waste) has become a topic of increasing concern in recent decades (Salehabadi, 2013) as stories of its mismanagement and illegal transboundary trade have emerged (Robinson, 2009). E-waste is not inherently damaging to the environment, however many electrical and electronic products contain a bewildering array of components, some of which contain potentially hazardous substances and materials (Townsend, 2011). If appropriate risk measures are not put in place to manage the deconstruction, reuse, recycling, recovery or disposal of 'after-use' items and their components, then potentially hazardous substances and materials may be at risk of emission into environmental media and/or human receptors (Tsydenova and Bengtsson, 2011).

The treatment and reclamation of valuable components (parts), materials and chemical compounds from e-waste in low income and middle income countries (LIMICs) is often unregulated, and informal (Duan et al., 2013; Tsydenova and Bengtsson, 2011). Therefore, it is challenging to determine the number of people involved, and the mass of material being processed, because informal and unregulated businesses inherently avoid or have no reason to report their activities (Nnorom and Osibanjo, 2008b). Once discarded, e-waste flows through society via complex and sometimes geographically diverse pathways, some of which may cross international boundaries as summarised in **Figure 1**. In the event that e-waste is collected for reclamation, it is likely to be processed via three broad groups of treatment types (Kaya, 2016; Ongondo et al., 2011; Tsydenova and Bengtsson, 2011):

(1) Physical processing: including the manual disassembly or comminution (size reduction) to recover components and materials.

(2) Hydrometallurgical treatment: whereby leaching solutions (lixiviant) such as nitric acid, hydrochloric acid or cyanide are used to dissolve metals bonded to electrical and electronic assemblies after which they are recovered from the solution through a variety of methods including precipitation, electrowinning and solvent extraction.

(3) Thermal processing: including the heating or combustion of electrical and electronic assemblies to melt solder or plastic bonding agents and housings to recover components and metals.

Each of these processes is controllable with sufficient safeguards in place. However, in the absence of regulation, enforcement and a positive health and safety culture (Glendon and Stanton, 2000), these processes may pose significant risk to those who are engaged in the

activities, resulting in the release of a range of potentially hazardous substances putting at risk of exposure human beings and/or environmental media.

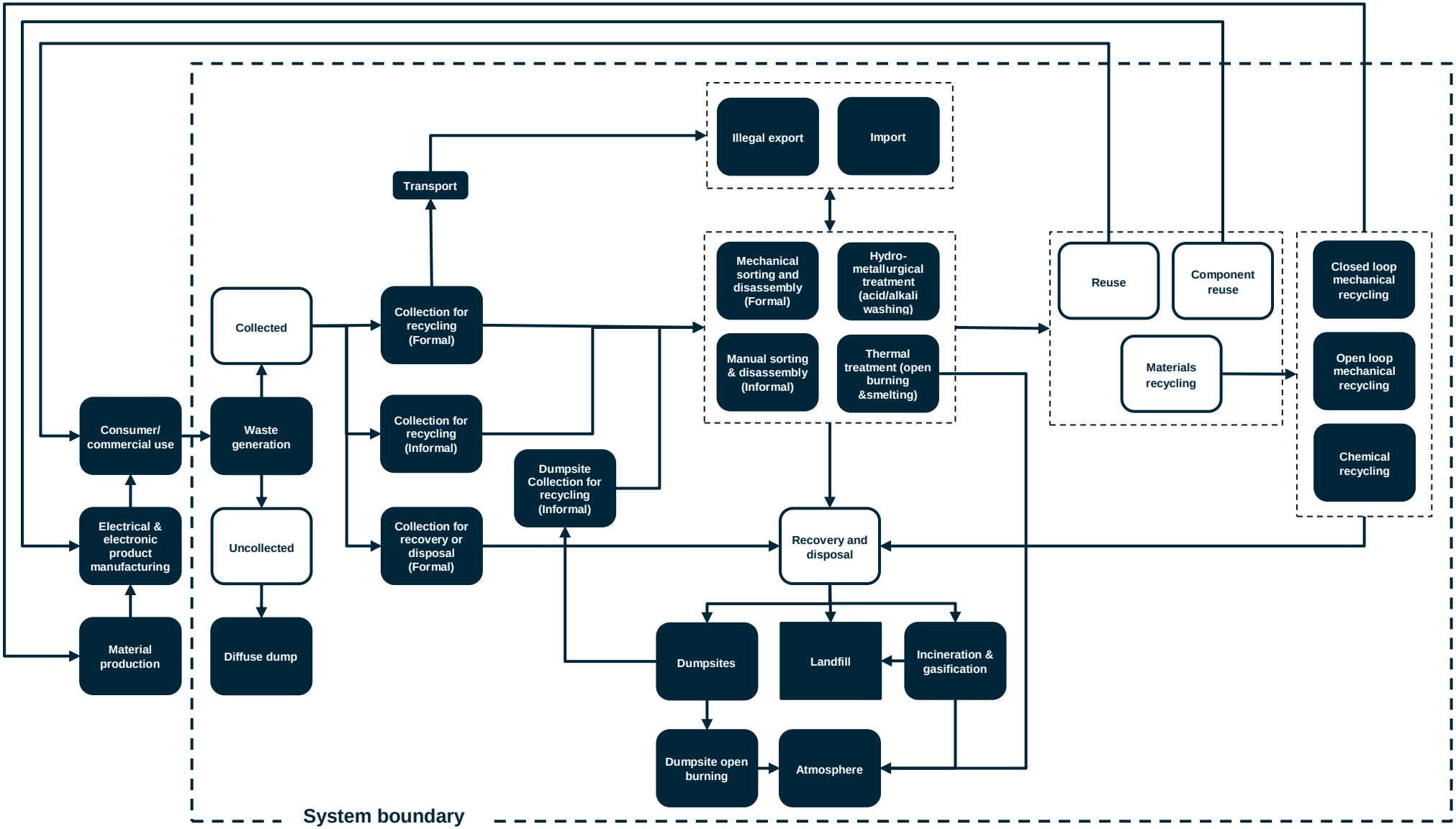


Figure 1: Material flow system for e-waste management through society.

In response to prior transgressions, the Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and their Disposal (hereafter the Basel Convention) has prohibited its parties from exporting broken electrical equipment from Organisation for Economic Co-operation and Development (OECD) countries to non–OECD countries since 2002 (Secretariat of the Basel Convention, 2002). The objective of this ban, is to prevent high income countries (HIC) outsourcing e-waste treatment to countries that have limited regulation and enforcement to ensure they are managed responsibly. Whilst this has been effective at curbing the legal trade in e-waste, there are concerns, and mounting evidence that the trade has been pushed underground (Hopson and Puckett, 2016) and many exporters now exploit a loophole whereby items are exported as functional used goods rather than waste, thereby potentially avoiding the gaze of inspectors (Lee et al., 2018; Lepawsky and McNabb, 2010).

The topic of e-waste is well researched in the literature compared to other waste materials such as construction and demolition waste (Cook and Velis, 2020a) or medical waste (Cook et al., 2020c). Several articles over the last decade have reviewed practices carried out to reclaim and reprocess e-waste, including studies with a scope at global level (Ongondo et al., 2011), regional level - Africa (Bimir, 2020) and Asia (Herat and Agamuthu, 2012), and socioeconomic level - developing countries (Nnorom and Osibanjo, 2008b). A recent and very detailed study by Kaya (2016) provides the most comprehensive overview of physical and chemical e-waste reclamation practices, but without any detailed evidence for the potential or realised health effects of the practices described. Other studies have provided more general reviews of e-waste policy, flow, and practices alongside assessment of solutions and prospects for long-term management of e-waste in various geographical areas including: Asian countries (Herat and Agamuthu, 2012), Pakistan (Iqbal et al., 2015), India (Borthakur and Govind, 2018), China (Lu et al., 2015), India and China (Awasthi and Li, 2017), Bangladesh (Bruce-Vanderpuije et al., 2019), Malaysia (Ismail and Hanafiah, 2019), Botswana (Mmereki et al., 2015), Ghana (Daum et al., 2017), and one global study with a Nigerian focus by Nnorom and Osibanjo (2008a).

There are also many primary data gathering studies relating to specific substances of concern emitted from e-waste processing activities, and some of these have been consolidated for groups such as for neurotoxicants (Chen et al., 2010), organic flame retardants (Gravel et al.,

2019a) and tetrabromobisphenol-A (TBBPA) (Malkoske et al., 2016). More comprehensive reviews have consolidated evidence for substance emission from e-waste activities and described the pathways through which receptors may be exposed. With the exception of a global review by Townsend (2011), these have had a regional, national or socio-economic category scope, for instance for: China (Xu et al., 2015), India (Awasthi et al., 2018; Awasthi et al., 2016), India and China (Brigden et al., 2005), Ghana (Awere et al., 2020; Brigden et al., 2008; Bruce-Vanderpuije et al., 2019), and for developing countries (Ackah, 2017).

Three studies have reviewed the impact of e-waste on public and occupational safety. Tsydenova and Bengtsson (2011) comprehensively summarised the state of knowledge for chemical hazard emissions from e-waste processing activities for 'developed' and 'developing' countries, providing extensive narrative to describe the link between emissions and processes in each of the two socio-economic groups. Grant et al. (2013), provided a thorough review of epidemiological studies investigating the association between e-waste and a range of health effects including physical health, neurodevelopmental disorders, violence, criminality and education. In a more recent study, Vaccari et al. (2019) systematically reviewed the impact of e-waste on environmental pollution and human health with a focus on the informal sector, listing and describing the current state of research with a focus on hazardous substance emissions and identification of substances in environmental media including a particularly strong focus on potentially toxic elements, with further reference to evidence for environmental concentrations of polybrominated diphenyl ethers (PBDEs), polychlorinated biphenyls (PCBs) and dioxin like PCBs, polychlorinated dibenzo-p-dioxins and polychlorinated dibenzo-p-furans (PCDD/Fs). Vaccari et al. (2019) was also able to compare emissions of various metals from different e-waste activities such as open burning, hydrometallurgical treatment and physical processing, finding a non-statistical inferred correlation between open burning and metal concentration in soils and between physical processing and higher atmospheric concentrations.

Whilst Tsydenova and Bengtsson (2011), Grant et al. (2013) and Vaccari et al. (2019) have provided comprehensive contributions the first was not a systematic review, and there has been a great deal of new research since publication of Tsydenova and Bengtsson (2011); and Grant et al. (2013), and the last does not provide detail on hazard exposure or quantified risk. To address this gap following a systematic and structured approach we have adapted

preferred reporting items for systematic reviews and meta-analyses (PRISMA) guidelines to identify, select and review sources that evidence occupational and public health risks associated with the processing of e-waste across global geographies and socio-economic cultures. Initially, we assessed the three broad groups of activities: 1) Physical processing, 2) Hydrometallurgical treatment and 3) Thermal processing. However, for the sake of clarity and brevity, we have chosen to present here only two of these thematic areas, described as Challenge 1: Physical processing (**Section 2.1**) and Challenge 2: Hydrometallurgical treatment (**Section 4**), leaving the thermal treatment for presentation in another publication (Cook et al., 2020b). For each Challenge, we summarise evidence for hazards emission, followed by sections that present the evidence of quantified risk of non-carcinogenic and carcinogenic effects.

Whereas our review is global, a strong focus is placed on activities in LIMICs where the most concerning safety challenges are evident. Deliberately, we have excluded from our scope, specific articles that cover lithium batteries, a new and specialist subject for which there are already two very recent reviews (Asadi Dalini et al., 2020; Siqi et al., 2019).

2. Methods

2.1. Systematic review

This study explored three research questions (**RQ**) via a systematic review PRISMA guidelines (Moher et al., 2009), adapted as detailed by Cook et al. (2020a):

- **RQ1:** What evidence exists to indicate risk to public and occupational safety posed by the physical and hydrometallurgical processing of e-waste?
- **RQ2:** What are the comparative risks to public and occupational safety that arise from the physical and hydrometallurgical processing of e-waste?
- **RQ3:** What research could be carried out that would have the greatest impact on harm reduction in the sector that processes e-waste using physical and hydrometallurgical processes?

Scopus, Web of Science and Google Scholar were searched using Boolean search queries that were optimised using one-at-a-time sensitivity analysis to ensure that the maximum number

of relevant articles were returned without an overburden of non-relevant articles (**Section S.1.1)**. Further supplementary search was carried out using snowballing and citation searching techniques (Cooper et al., 2018). Literature was screened according to the inclusion and exclusion criteria detailed in **Section S.1.2** to achieve the results detailed in **Section S.1.3.** Further sources were obtained by searching through the websites of international and national organisations such as the United States Environmental Protection Agency (2020), Occupational Safety and Health Administration (2020), The World Bank (2020) and World Health Organization (2020).

Risks, hazards, pathways and receptors were identified in literature and combined into scenarios that were either reported directly or could be conceived to be feasible from the descriptions in each source as described by Cook et al. (2020a). These combinations enabled the production of conceptual diagrams as shown in **Figure 2**.

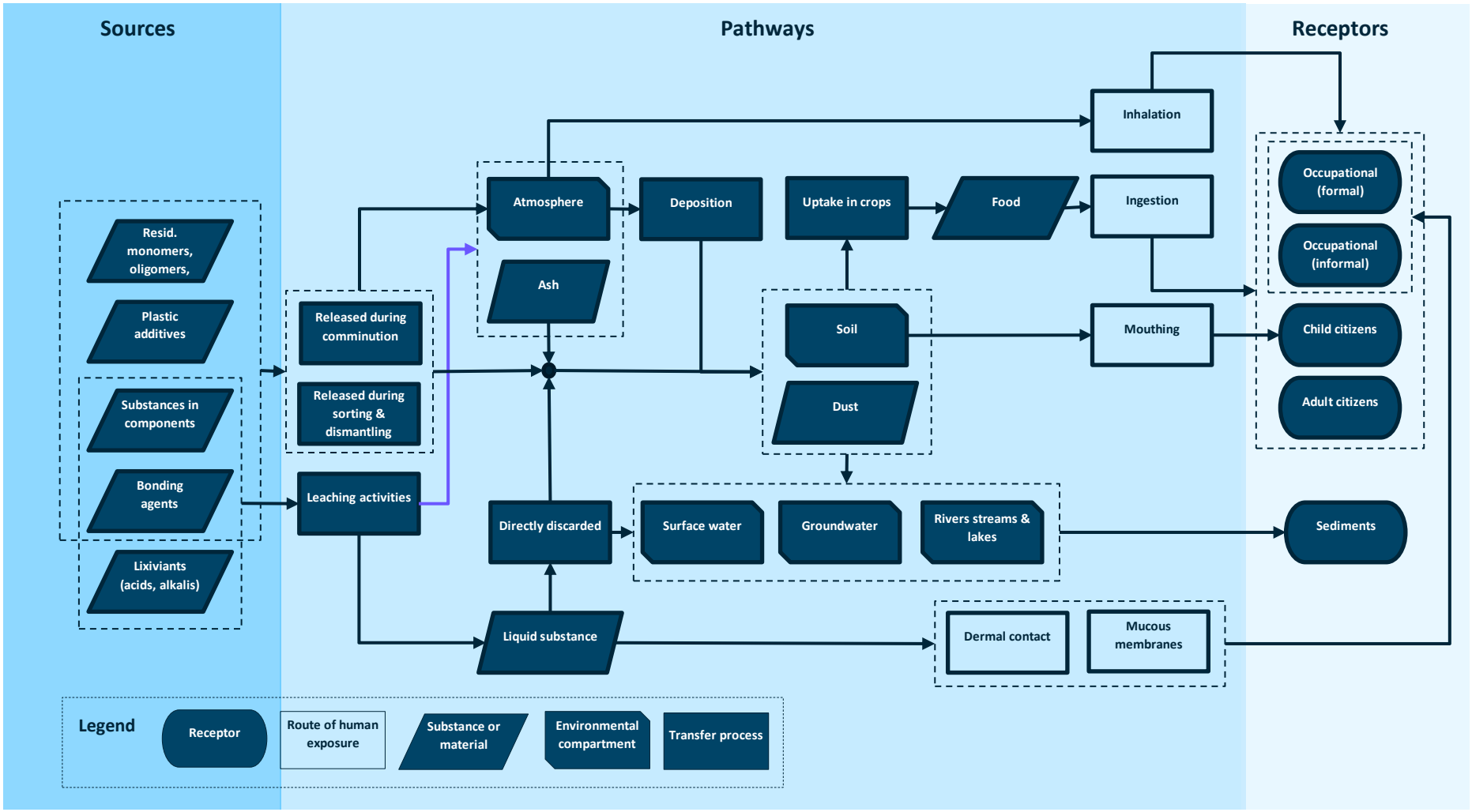


Figure 2: Summary of the main sources, pathways and receptors for hazards associated with e-waste (purple arrow delineates crossing arrow and has no further meaning).

2.1. Risk based approach

In order to rank and compare the relative risks, an approach adapted from Kaya et al. (2018), Burns et al. (2019), World Health Organization (2012), and Hunter et al. (2003) was used to indicatively score each of the hazard-pathway-receptor combinations on the basis of the likelihood and severity of each occurring using the matrices in **Table 1** and **Table 2**. It is important to note that this method, reported first by Cook et al. (2020a), was not intended to constitute or substitute for a quantitative assessment of risk, but to be used to support decision making by indicating the relative harm from each scenario and assist with directing future research agenda. The ranked, aggregated results are shown in **Section S.2**.

Table 1: Matrix used to calculate the relative risk of each hazard-pathway-receptor scenario; after Cook et al. (2020a).

			Consequence				
			Very slight	Slight	Moderate	Severe	Very severe
			1	2	3	4	5
Likelihood	Very unlikely	1	1	2	3	4	5
	Unlikely	2	2	4	6	8	10
	Likely	3	3	6	9	12	15
	Very likely	4	4	8	12	16	20
	Inevitable	5	5	10	15	20	25

Table 2: Colour coding used to rank hazard potential qualitatively in each category; after Cook et al. (2020a).

Red (R)	High harm potential
Amber (A)	Medium/high harm potential
Yellow (Y)	Medium/low harm potential
Green (G)	Low harm potential
Grey	Insufficient data

3. Challenge 1: Sorting, dismantling and physical processing of e-waste

E-waste that has been collected for recycling or refurbishment/reuse is often subject to a multiplicity of processes that are intended to maximise the value from the constituent product components, each of which may contain many subcomponents, substances and materials (Tzoraki et al., 2019). E-waste has a varying material and chemical composition, which includes a variety of potentially hazardous substances and materials that may negatively affect the health of receptors if a pathway is created.

In the following sections, evidence is reviewed for emissions of potentially hazardous substances from the handling, dismantling and physical processing of e-waste. Comparisons are made between the concentrations of substances present in occupational environments in which the formal and informal sector operate, as well as the levels observed in nearby environmental compartments and the blood urine and hair of those who participate in e-waste dismantling activities.

The potential hazards posed by e-waste sorting, dismantling and physical processing are simplified in **Figure 3**. In isolation, these hazards pose no threat to receptors, unless one of the pathways illustrated allows them to encounter them.

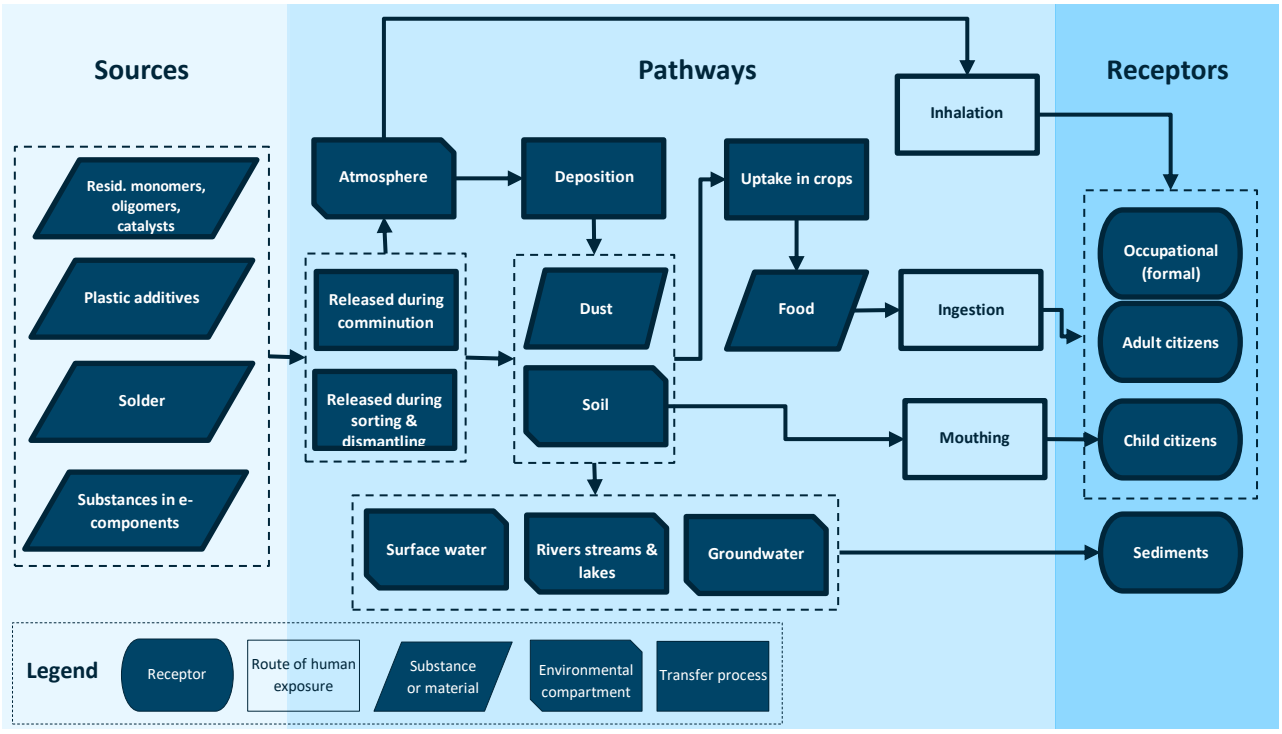


Figure 3: Hazard exposure conceptual model (source–pathway–receptor) associated with sorting, dismantling and physical processing of e-waste.

3.1. Potentially toxic elements (PTEs)

In the context of interaction with biota, metals can be categorised as those which are essential for organisms to function and those which are not (Egorova and Ananikov, 2017). Popular sentiment incorrectly assumes that the latter group are those which are more likely to result in harmful effects, however metals that are essential for life may also be toxic if the dose is sufficient. In this section, studies are compared that report the concentration of metals and

other elements as they have been observed in the human body and several environmental compartments. These elements are typically described in literature and throughout as ‘heavy metals’; a non-specific category and one which excludes metalloid and non-metal elements. Instead, the present study will use the term potentially toxic elements (PTEs), following the recommendation of Pourret and Hursthouse (2019).

The present research identified 22 studies that provided evidence for PTE emissions from e-waste recycling and treatment activities carried out by the formal and informal sector. Of these, several provided specific data, or an inference, that the concentrations of PTEs identified in people or environmental compartments originated as a consequence of sorting, dismantling and physical processing. Of course, other activities are carried out by e-waste recyclers which result in emissions of PTEs, such as open burning, heating and smelting to recover solder and components as described by Cook et al. (2020b) and hydrometallurgy to recover metals as described in **Section 4**. It is therefore challenging to disaggregate emissions from these sources as they often take place in the same locations. Nonetheless, it is important to understand the contribution of specific practices to emissions of potentially hazardous substances so that interventions can be implemented efficiently and that the most damaging activities receive attention first.

E-waste dismantling in HICs make a useful context for comparison with studies in a LIMIC context, as practices such as open burning are unlikely to take place in HICs and heating and hydrometallurgy are likely to be carried out with engineering control measures in place, such as local exhaust ventilation. Five studies summarised in **Table 3** measured atmospheric PTE concentrations in or around e-waste recycling areas in different contexts, two of which were in HICs.

Table 3: Selected elements detected in air samples (μg m^{-3}) close to e-waste dismantling, sorting and physical processing sites.

Ref.	Geog.	Receptor	Activity context	n	Pb	Cd	Cr	Cu	Mn	Mo	In	Hg
			All activities	77[f]	7[i]	0.18[i]	0.45[i]	2.2[i]	2.2[i]	0.050[i]	0.018	0.011[i]
			Dismantling	34[f]	8[i]	0.3[g i]	0.58[i]	2.9[g i]		0.064[g]	0.26[g h]	0.012[i]
		Formal recycling workers	Indoors	30[f]	6.8[i]	0.15[i]	0.43[i]	2[i]		0.050[i]	0.018[i]	0.012[i]
			Outdoors	13[f]	5.2[i]	0.027[i]	0.27[i]	1.3[i]		0.002[i]	0.070[i]	0.0001[i]
Julander et al. (2014)	SWE	Office workers	Office	3[f]	0.4	0.0053	0.036	0.093	0.11	0.003	0.0004	0.00039
			Shredder sorting (A)	9	1.6-67[†*¥]	ND-0.84						
			CRT buffing and grinding (A)	5	9.8-27	0.18-10[†¥]						
			CRT operators (A)	6	6.1-16	0.09-0.34						
			CRT dismantling (A)	8	2.1-5.3	ND-0.84						
			Baler, battery bulb sorting (A)	5	0.9-3.9	ND-0.065						
			Shipping, receiving, shredding (C)	12	0.33-3.3	ND-0.08						
		Formal recycling workers	Battery sorters (C)	13	0.2-3.3	ND-0.065						
Ceballos et al. (2017)	USA		Dismantling (C)	9	0.37-1.2	ND-0.14						
Huang et al. (2016)	CHN	Informal recyclers	Recycling[b]	165	0.17	0.006	0.023	0.09	0.076	0.003		
			Recycling	300[a]	0.153[c]	0.006	0.006		0.022			
Zeng et al. (2016)	CHN	Informal recyclers	Ref.	170[a]	0.080[c]	0.003	0.008		0.023			
		Informal recyclers	Recycling	15								0.0307[d]
			Village 3 km[e] from recycling area	15								0.0154[d]
Tang et al. (2015)	CHN	Residents	Ref.	15								0.0072

Air concentrations benchmarked against UK and US occupational exposure limits (**Section S.4.1**) except for ACGIH® 2019 TLV® time weighted averages which are proprietary, as follows: [♯] exceeds eight-hour time weighted average (TWA) indoor air reference value set by the UK Health and Safety Executive (HSE); [‡] exceeds eight-hour TWA permissible exposure limit set by US OSHA; [†] exceeds eight-hour TWA permissible exposure limit set by Californian OSHA; * exceeds 10-hour TWA recommended exposure limit set by NIOSH (US); [¥] exceeds eight-hour TWA threshold limit value set by the ACGIH® 2019 TLV®; [a] In PM2.5 air samples; [b] average of three heights above the ground in e-recycling zone; [c] significant difference between values using T-test based on Ln-transform ($p < 0.000$); [d] significant difference between values using Pearson's correlation coefficient ($p > 0.05$); [e] village was 3 km distance from an area where significant recycling of e-waste was taking place; [f] inhalable fraction; [g] dismantling workers significantly more exposed compared to outdoor workers ($p < 0.05$); [h] dismantling workers significantly more exposed compared to indoor workers ($p < 0.01$); [i] dismantling workers significantly more exposed compared to office workers ($p < 0.05$). (A) and (C) denote different facilities studied by Ceballos et al. Abbreviations: Geographical context (Geog.); number of samples (n); dismantling (dis.); recycling (rec.); cathode ray tube (CRT); reference (Ref.); not detected (ND).

The first, Julander et al. (2014), analysed the indoor and outdoor atmosphere at three e-waste recycling facilities in Sweden, comparing the exposure potential between workers carrying out different activities. Although the concentration of the PTEs studied did not exceed guideline limits from any of the UK or US safety institutions, significantly higher concentrations of cadmium (Cd), copper (Cu), molybdenum (Mo) and indium (In) were experienced by dismantlers in comparison to the workers whose activities were outdoors.

In another HIC context, Ceballos et al. (2017) measured Pb and Cd concentrations in air at three e-waste recycling plants in the US, finding comparatively low concentrations in most samples with three exceptions. One out of the nine samples (specific data not shown) collected from workers in the shredding and sorting area showed a concentration of Pb that exceeded US guidelines **(Section** Error! Reference source not found.**S.4.1)**. Further, two out of five samples collected from cathode ray tube (CRT) buffing and grinding exceeded US guidelines for Cd concentrations in air. It is noteworthy that none of the samples for lead (Pb) or Cd exceeded the UK guidelines. Occupational exposure limits in the UK and other parts of Europe have been shown to be higher for comparable substances compared to the US or Australia for instance Schenk et al. (2008), reflecting different attitudes and interpretation of evidence in different societal attitudes towards risk. Whereas it is beyond the scope of the present research to compare guidelines of different national agencies, the fact that the levels identified fell between the two limits and not above, provides an inference that the concentrations were unlikely to result in serious adverse effects.

Concentrations of PTEs measured at e-waste dismantling plants in China reported by Huang et al. (2016), Zeng et al. (2016) and Tang et al. (2015) were all below US and UK limits (**Table 3**). While this may be unexpected, it is suggested here that the level of mechanisation is much greater at the HIC context facilities, increasing the likelihood of particle transmission to the atmosphere.

The differences between exposed and reference groups for Pb and mercury (Hg) concentrations measured in air reported by Zeng et al. (2016), Tang et al. (2015) and Julander et al. (2014), were all significant and for Julander et al. (2014), they were significant for all other metals except for In. Whereas the results of the studies by Tang et al. (2015) and Zeng et al. (2016) may be confounded by potential, open burning activities in the sampling area,

this is unlikely to be the case for the facilities investigated by J Julander et al. (2014), providing a justifiable link between elemental releases and physical, non-thermal processing.

One other study by Fang et al. (2013) reported concentrations of PTEs in mg g^{-1} of particulate matter (PM) making it non-comparable with the other studies (**Table 4**). The difference between concentrations in the two workshops were not compared statistically, however approximately two orders of magnitude more Pb can be observed in the dismantling workshop than the mechanical workshop, approximately 3–4 times more Cd, twice the concentration of Cr, a similar quantity of Cu and 1,000 times more Ni.

Table 4: Element concentrations in air samples at e-waste dismantling workshops in Shanghai, China observed by Fang et al. (2013).

Activity location	Units	n	Pb	Cd	Cr	Cu	Ni
Mechanical workshop	mg g^{-1} PM_{10}	2x3	12.34	0.108	0.554	27.76	0.472
	mg g^{-1} $PM_{2.5}$	2x3	20.46	0.033	1,202	3,753	0.744
Dismantling workshop	mg g^{-1} PM_{10}	2x3	2,043	0.398	0.436	31.80	0.459
	mg g^{-1} $PM_{2.5}$	2x3	6,935	0.094	2,875	1,205	1,148

Abbreviations: number of samples (n); particulate matter < 10 µm (PM_{10}); particulate matter < 2.5 µm ($PM_{2.5}$).

3.1.1. Soil and dust

In addition to measuring atmospheric concentrations of PTEs at dismantling, sorting and physical processing plants, investigations into soil and dust concentrations also provide evidence that indicates the potential hazard exposure. Nine studies reviewed here reported concentrations of 18 PTEs (**Table 5** and **Table 6**) at formal and informal e-waste processing facilities in seven countries. For context, each reported value has been benchmarked against United States Environmental Protection Agency (USEPA) Soil Screening Levels (United States Environmental Protection Agency, 2019b) (**Section S.4.2**). While these levels provide an indication as to the level of contamination, the reported values have not been put into context with background values, which may partly or entirely explain the presence of the various PTEs identified.

Table 5: Element concentrations observed in soil in and around e-waste dismantling, sorting and physical processing plants mg kg^{-1}).

Ref.	Geog.	Activity context	Media	n	Ag	As	Cd	Co	Cr	Cu	Fe	Ga [g]	Hg
		E-recycling site (formal)		6 [a b]	1.5	2.6[†]	0.3	21[¥]		350[¥]	31,000[¥]		
		E-recycling site (informal)	Soil	7 [a b]	4	3.2[†‡]	2.9	26[¥]		810[¥]	32,000[¥]		
		E-recycling site (formal)		11 [a]	180[¥]	5.1[†¥‡]	2.9	53[¥♯]		23,000[¥♯]	42,000[¥]		
Fujimori et al. (2012)	PHL	E-recycling site (informal)	Dust	6 [a]	130[¥]	7.6[†¥‡]	3	17[¥]		6,600[¥♯]	52,000[¥]		
		Dis. area (D season)		10 [c]			10.29[¥]		36.78[†¥‡]	3,165[¥]			
Isimekhai et al. (2017)	NGA	Dis. area (W season)	Soil	10 [c]			8.67[¥]		49.6[†¥‡]	5,580[¥♯]			
				41 [b]	16	271[†¥‡♯]	11[¥]		103[†¥‡]	11,200[¥♯]	56,800[¥]		
Ackah (2019)	GHA	Dis. Area	Soil	37 [d]	12	5[†¥‡]	8[¥]		60[†¥‡]	1,800[¥]	33,400[¥]		
		Dis. area		29 [b]	80[¥]	25[†¥‡]	2.4	3.4[¥]	118[†¥‡]	3,199[¥]	34,140[¥]	12.4	10.2[¥♯]
		Repair area		29 [b]	1	3.6[†¥‡]	2.6	1.5	49[†¥‡]	28	11,000[¥]	4	0.5
				29 [b]	19	22[†¥‡]	5	15.5[¥]	117[†¥‡]	3,399[¥]	62,896[¥]	9	6[¥♯]
			Soil	29 [b]	15	40[†¥‡]	24[¥]	1.5	197[†¥‡]	7,880[¥♯]	70,090[¥]	14.5	0.5
				32	38.5	8.4[†¥‡]	2.6	7[¥]	36[†¥‡]	2,062[¥]	30,620[¥]	8.6	3.3[¥]
		Dis. area	Dust	32	40[¥]	71[†¥‡]	30[¥]	1.5	188[†¥‡]	13,580[¥♯]	97,260[¥♯]	15	0.5
		Dis. area (control soil)	Floor dust	n/a	34.5	5.7[†¥‡]	2.55	1.5	78[†¥‡]	766[¥]		7.8	2.3[¥]
		Dis. sites	Roadside dust	n/a	7.7	13[†¥‡]	2.55	2.8[¥]	62.5[†¥‡]	234		5.8	0.18
		Dis. sites	Direct electronic dust	n/a	1.35	1.2[†]	2.55	72.3[¥♯]	14.3[†¥‡]	42		21	0.5
Ohajinwa et al. (2018)	NGA	Dis. area (control soil)	Floor dust	n/a	40[¥]	71[†¥‡♯]	30[¥]	1.5	188[†¥‡]	13,580[¥♯]		15	0.5
		E-recycling site		15 [e]									3.1[¥]
		Village 3 km away		15 [e]									1.3[¥]
		Ref. area	Soil	15 [e]									0.1
Tang et al. (2015)	CHN	E-recycling site	Dust	15									37.6[¥♯]
		Adjacent residential		4 [f]						58-143	16,285-19,124[¥]		
Damrongsiri et al. (2016)	THA	Dis. area	Soil	11 [f]						214-12,986[¥♯]	26,968-102,580[¥♯]		
		Dis. w'shop		5		22[†¥‡]	3	7[¥]	63[†¥‡]	273			3[¥]
Chakraborty et al. (2019)	IND	Sorting w'shop	Soil	4		0.2	1	3[¥]	110[†¥‡]	533[¥]			1
Tzoraki et al. (2019)	GRC	Around the plant	Soil	25	0.057-0.721	6.7-21.7[†¥‡]	0.13-2.42	4.2-15.4[¥]	21.9-114.4[†¥‡]	20.62-254.23		0.9-4.2	0.033-0.238
		Floor dust mech. w'shop		n/a			92[¥]		174[†¥‡]	947[¥]			
Fang et al. (2013)	CHN	Floor dust dis. w'shop	Dust	n/a			59[¥]		152[†¥‡]	2,160[¥]			

[a] mg/kg air-dry-base; [b] depth 030 cm; [c] depth 0-10 cm; [d] depth 30-100 cm; [e] depth 5-10 cm; [f] depth 10 cm; [g] no guidelines were published by USEPA and hence not benchmarked; [†] exceeds USEPA carcinogenic screening level (TR=1E-06) for residential soils; [¥] exceeds USEPA child non-carcinogenic screening level (THI=0.1) for residential soils (**Section S.4.2**); [‡] exceeds USEPA carcinogenic screening level (TR=1E-06) for industrial soils; [♯] exceeds USEPA non-carcinogenic screening level (THI=0.1) for industrial soils; specific screening level elements were compared to: Cd to 'cadmium (Diet)'; Cr to 'chromium (VI)'; Hg to 'elemental mercury'. Abbreviations: Geographical context (Geog.); number of samples (n); dismantling (dis.); mechanical (mech.); not available (n/a)

Table 6: Element concentrations observed in soil in and around e-waste dismantling, sorting and physical processing plants (mg kg-1).

Ref.	Geog.	Activity context	Media	n	In	Mn	Mo	Ni	Pb	Sb	Se	V	Zn
		E-recycling site (formal)		6 [a b]	1	800[†]		16	53				140
		E-recycling site (informal)	Soil	7 [a b]	<0.5	900[†]		64	650[¥]				1,000
		E-recycling site (formal)		11 [a]	19	820[†]		2,600[¥♯]	9,000[¥♯]				3,000[¥]
Fujimori et al. (2012)	PHL	E-recycling site (informal)	Dust	6 [a]	<0.5	2,100[†]		200[¥]	1,400[¥♯]				2,800[¥]
		Dis. area (D season)		10 [c]		254.9[†]		77.4	911[¥♯]	22.5[¥]			862.6
Isimekhai et al. (2017)	NGA	Dis. area (W season)	Soil	10 [c]		120.8		23.91	1,823[¥♯]	58.4[¥♯]			1,921
				41 [b]					2,380[¥♯]		5		1,820
Ackah (2019)	GHA	Dis. area	Soil	37 [d]					846[¥♯]		3		866
		Dis. area		29 [b]		672[†]	5.6	103[¥]	1,723[¥♯]	523[¥♯]	2.4	51[¥]	2,534[¥]
		Repair area		29 [b]		81	1.5	100[¥]	12	1.5	0.3	10	25
				29 [b]		859[†]	6.5	95[¥]	11,757[¥♯]	78[¥♯]	14	52[¥]	2,724[¥]
			Soil	29 [b]		540[†]	17	153[¥]	2,271[¥♯]	204[¥♯]	12	46[¥]	5,650[¥]
				32		17,094[†♯]	2.1	131[¥]	370	84.5[¥♯]	0.7	36	1,616
		Dis. area	Dust	32		20,265[†♯]	18	149[¥]	3,770[¥♯]	382[¥♯]	4	26	5,401[¥]
		Dis. area (control soil)	Floor dust	n/a		680[†]	0.5	99.4[¥]	277	43[¥]	0.65		1,342
		Dis. sites	Roadside dust	n/a		426[†]	0.5	83	374	14[¥]	0.6		924
		Dis. sites	Direct electronic dust	n/a		1,607[†]	1.3	320[¥]	0.5	1.5	0.25		122
Ohajinwa et al. (2018)	NGA	Dis. area (control soil)	Floor dust	n/a		654[†]	17.5	149[¥]	3,770[¥♯]	382[¥♯]	4		5,401[¥]
		Adjacent residential		4 [f]		350-405[†]		12-23	40-92				158-316
Damrongsiri et al. (2016)	THA	Dis. area	Soil	11 [f]		466-674[†]		16-183[¥]	86-4,556[¥♯]				182-4,258[¥]
		Dis. w'shop		5				75	197				
Chakraborty et al. (2019)	IND	Sorting w'shop	Soil	4				234[¥]	87				
Tzoraki et al. (2019)	GRC	Around the plant	Soil	25	<0.02-0.03	201-972[†]	0.27-1.72	14.2-99.7[¥]	<0.010-0.083	0.17-45.42[¥]	<0.1-0.5	2-55[¥]	30.9-1,089.2
		Floor dust mech. w'shop		n/a				1,225,000[†¥‡♯]	13,880[¥♯]				
Fang et al. (2013)	CHN	Floor dust dis. w'shop	Dust	n/a				318[¥]	17,830[¥♯]				

[a] mg/kg air-dry-base; [b] depth 0-30 cm; [c] depth 0-10 cm; [d] depth 30-100 cm; [e] depth 5-10 cm; [f] depth 10 cm; [†] exceeds USEPA carcinogenic screening level (TR=1E-06) for residential soils (**Section S.4.2**); [¥] exceeds USEPA child non-carcinogenic screening level (THI=0.1) for residential soils; [‡] exceeds USEPA carcinogenic screening level (TR=1E-06) for industrial soils; [♯] exceeds USEPA non-carcinogenic screening level (THI=0.1) for industrial soils; specific screening levels elements were compared to: Mn as Manganese 'non-dietary'; Ni as 'Nickel oxide'; Sb as metallic antimony. Abbreviations: Geographical context (Geog.); number of samples (n); dismantling (dis.); mechanical (mech); not available (n/a).

Many of the values reported exceeded Soil Screening Levels set for residential soils which are set to the lowest common denominator that assumes that children will ingest soil. While several sources reviewed provide evidence that children live on e-waste sites, it is possibly more relevant to consider the concentrations that exceed the levels set for industrial premises which are deliberately set higher as it is considered that workers are less exposed to soils in comparison to a domestic context. Two elements, As and Cr, exceeded carcinogenic industrial soil screening level at 16 and 18 sites respectively. Pb was also prevalent, exceeding the non-carcinogenic industrial level at 14 sites.

Two sites reported by Fang et al. (2013) showed very high levels of nickel (Ni) and Pb in dust collected from the mechanical workshop floors of e-waste dismantling plants in Shanghai, China. Levels of Ni here were more than 1,000 times the non-carcinogenic industrial soil screening level set by USEPA and levels of Pb were 17 and 22 times the level. Other high levels of Pb were reported by Fujimori et al. (2012) who observed levels 11 times the non-carcinogenic industrial soil screening level set by USEPA in formal e-waste recycling sites in the Philippines where levels were higher than for the informal sector sites investigated.

3.1.2. Water

PTEs have also been observed in drinking and environmental freshwater in and around e-waste dismantling facilities (**Table 7**). Tang et al. (2015) observed Hg and Zhang et al. (2019) observed Cd, Cu, Pb and zinc (Zn) all at concentrations below USEPA guidelines for tap water, freshwater aquatic life and freshwater human consumption (**Section S.4.3**). Tzoraki et al. (2019) analysed well water, stream water and water from a storage tank at an e-waste dismantling site, finding values generally below thresholds for drinking water safety. A notable exception was one sample which exceeded the limit for arsenic (As), possibly because of local volcanic activity rather than the e-waste dismantling activities. Several other samples exceeded limits for drinking water by a small margin for some elements, notably Cd, cobalt (Co), chromium (Cr), Cu, Pb, manganese (Mn), antimony (Sb), and thallium (Tl). Chronic freshwater (aquatic life) limits were exceeded for Pb in 50% of cases, with one exceeding by 13 times, and Cd in just one case out of eight by a small margin. The paper found a statistically significant correlation between the company's activities and the concentrations on the basis of elemental composition of the processed feedstock, indicating a clear link between its activities and local pollution of the soil and water.

Table 7: Elemental concentrations observed in water in and around e-waste dismantling sites (µg L^{-1}).

	Tang et al. (2015)	Zhang et al. (2019)		Tzoraki et al. (2019)	
	Residents in e-waste recycling (CHN)	Residents in dismantling area (CHN)	Ref. area (CHN)	Residents and workers (GRE)	
Analyte	Drinking water (n=15)	Drinking water (n=25)	Drinking water (n=3)	Streams (n=6) Storage tank (n=1) Well (n=1)	
Ag				<0.05-0.11	
Al				24-717	
As				2.9-12.2	¥†∗₸
Be				<0.05-0.12	
Cd		0.048	<0.01	<0.05-1.02	¥♯
Co				0.09-1.23	¥
Cr				0.07-10.4	¥†
Cu		66.5	65.3	3.5-81.1	¥
Ga				<0.05-0.19	
Hg	0.014			-	
Mn				3.97-177	¥
Mo				0.3-6	
Ni				0.2-19.2	
Pb		1.91	1.5	1.1-43.5	¥♯
Pt				<0.01-0.01	
Sb				0.15-2.91	¥
Se				<0.5-0.8	
Tl				<0.01-0.03	¥
W				0.06-0.26	
Zn		80.7	75.1	4.9-328.2	

† Exceeds USEPA carcinogenic screening level (TR=1E-06) for residential drinking water (**Section S.4.3**); ¥ exceeds USEPA child non-carcinogenic screening level (THI=0.1) for residential drinking water; ‡ exceeds USEPA acute quality criteria for freshwater aquatic life; ♯ exceeds USEPA chronic quality criteria for freshwater aquatic life; * exceeds USEPA quality criteria for freshwater consumption of water and organisms; ₸ exceeds USEPA quality criteria for freshwater consumption of organisms only. Abbreviations: number of samples (n); reference (Ref.)

3.1.3. Food

Two authors also reported concentrations of PTEs in foodstuffs near to e-waste dismantling areas in China (**Table 8**). The analysis by Tang et al. (2015) tested samples for Hg finding that they exceeded thresholds set by the Chinese National Standard Agency in just a few cases. Limits are commonly provided for Hg in fish, which is thought to be a significant source of human intake. Therefore few guidelines are available to compare levels with other foodstuffs; however, none of the mean concentrations exceeded the threshold for fish set in European Commission Regulation 1881/2006 (European Union, 2006; European Union, 2015) (**Section S.4.4)**. Zhang et al. (2019) analysed samples for Pb, Cd, Cu and Zn, finding levels of Pb that equalled or exceeded by a small margin the limits stated in Commission Regulation 1881/2006, with higher concentrations detected in e-waste dismantling areas compared to the reference samples. Similarly, levels of Cd were notably higher in the e-waste dismantling sites, however none exceeded Commission Regulation thresholds.

Table 8: Elemental concentrations observed in food and crops (µg g^{-1} foodstuff) around e-waste dismantling sites.

Ref.	Geog.	Activity context	Food	n	Pb	Cd	Cu*	Zn*	Hg
			Rice*	15					0.0505
			Corn*	15					0.068
			Soybean*	15					0.043
			Cole*	15					0.025
			Grain*	15					0.056
			Rice (ref)*						0.012
			Chicken & pork*	15					0.0601
			Fish	15					0.2298
			Milk*	15					0.0029
			Food oil*	15					0.0054
Tang et al. (2015)	CHN	Village 3 km away	Table salt*	15					0.0045
		Residents living e-waste dismantling area	Vegetables	46	0.35†	0.096	1.37	3.89	
		Residents ref. area	Vegetables	4	0.1†	0.023	1.75	6.23	
		Residents living e-waste dismantling area	Rice	32	0.2†	0.15	12.3	27.7	
		Residents ref. area	Rice	5	0.18	0.041	9.38	24.6	
		Residents living e-waste dismantling area	Egg*	28	0.071	0.006	1.01	16.2	
Zhang et al. (2019)	CHN	Residents ref. area	Egg*	10	0.014	0.0002	0.63	12.2	

†Exceeds limits on concertation as per Commission Regulation 1881/2006 (European Union, 2006; European Union, 2015) (**Section S.4.4**); *no threshold comparison stated in Commission Regulation 1881/2006. Abbreviations: Geographical context (Geog.); number of samples (n); reference (ref.).

3.1.4. Human exposure

Whereas elemental concentrations in various environmental compartments are an indicator of environmental emissions, observations of these substances in blood, urine and hair evidence exposure and absorption, which can occur through a variety of mechanisms (exposure pathways). Many PTEs which have not accumulated in the body are excreted through urine, indicating exposure over recent hours or days (Zhang et al., 2019). Two studies, Zhang et al. (2019) and Julander et al. (2014) reported concentrations of elemental mass per volume of urine for e-waste dismantlers in China and Sweden respectively. In **Table 9**, the results of these studies are shown alongside concentrations of non-occupationally exposed UK residents' urine for comparison; though these should be treated with caution as dietary intake and environmental conditions can significantly influence concentrations of some elements (Asante et al., 2012).

Table 9: Element concentrations observed in urine of people working and living around e-waste dismantling facilities (μg L^{-1} urine).

Element	Zhang et al. (2019) (CHN)		Julander et al. (2014) (SWE)		Morton et al. (2014) (GBR)
					Non-occupationally exposed adults
	Residents near e-waste (n=139)	Residents ref. area (n=26)	E-recycling workers (n=52)	Office workers (n=10)	All (n=132)
Al	17	19.1			3.82
As	46.6	62	13	19	10.48
Cd	2.12*	1.33	0.37	0.27	0.13
Co	0.44	0.32	0.25	0.24	0.22
Cr			0.74	0.71	0.35
Cu	22.2	16.9			8.75
Hg	0.5	0.42	1.4	0.66	0.43
In			0.0068	0.0047	<LOQ (0.031)
Mn	0.77	0.61			<LOQ (0.092)
Pb	4.98***	1.23	1.8**	0.66	0.47
Sb	0.2	0.11			<LOQ (0.092)
Se	30.5	26.3			13.4
Tl	0.57	0.59			0.17
Zn	530	493			n/a

*Significantly higher ($p < 0.05$) in e-waste recycling area compared to reference (Mann–Whitney U-test); **significantly higher ($p < 0.01$) in e-waste recycling area compared to reference (Mann–Whitney U-test); ***significantly higher ($p < 0.001$) in e-waste recycling area compared to reference (Mann–Whitney U-test). Abbreviations: Number of samples (n); reference (ref.); limit of quantification (LOQ); not available (n/a).

Urinary concentrations of most elements were higher in the study of exposed and non-exposed Chinese residents compared to those in the UK or Sweden. An interesting exception is the concentrations of Hg in the urine of both groups of Chinese residents that were similar to UK non-exposed residents. In contrast, the levels of Hg in e-waste workers in Sweden were almost double in the e-waste workers and slightly higher in the office workers at the e-waste recycling facility. In the case of Cd and Pb, levels were significantly higher for e-waste workers in China compared to the reference group ($p < 0.05$ for Cd; $p < 0.001$ for Pb), clearly indicating occupational exposure to these elements.

A similar correlation between e-waste dismantling and recycling activities and exposure to PTEs is indicated by three studies that analysed the blood of people working in the sector (**Table 10**). For instance, Julander et al. (2014) observed concentrations of Cr and Pb to be higher in occupationally exposed workers compared to office workers in the same premises with a significance of $p < 0.05$ and $p < 0.01$ respectively.

Table 10: Blood concentrations of selected elements compared for adults working near and in e-waste recycling areas (μg L^{-1}).

Ref.	Geog.	Activity context	n	Cd	Co	Cr	Hg	In	Pb
			50		0.081	1.4†*	1.4	0.0057	32**
		E-recycling workers	48[a]			0.81†*		0.0043*	
			10		0.064	1.1†	1.2	0.0030	15
Julander et al. (2014)	SWE	Office workers	10[a]			0.30		0.0023	
		Baler, shredder, battery (A)	5	ND-12†					ND-88†
		CRT shredders (B)	7	ND-17†					ND-46†
		Dismantling (B)	13	ND-9†					ND-13
		Shipping, receiving, shredding (C)	5	5-10†					ND
		Dismantling (C)	3	6-15†					ND
Ceballos et al. (2017)	USA	Battery sorters (C)	2	5-6†					ND
Amankwaa et al. (2017)	GHA	E-waste dismantlers	28						36.3†

[a] As concentration in plasma/serum fraction; † exceeds reference value for blood on non-occupationally exposed Canadian adult citizens (20 to 79 years of age) **(Section S.4.5)**; ♯ exceeds reference value for blood on non-occupationally exposed Canadian citizens (3 to 79 years of age); * significantly higher ($p < 0.05$) in e-waste recycling area compared to reference (Mann–Whitney U-test); ** significantly higher ($p < 0.01$) in e-waste recycling area compared to reference (Mann–Whitney U-test). Abbreviations: Geographical context (Geog.); number of samples (n); cathode ray tube (CRT); Not detected (ND).

Ceballos et al. (2017) did not analyse blood from the control group or statistically compare variance between the workers, however comparison with average concentrations in the blood of Canadian adults between 20 and 79 years old (**Section S.4.5**) shows levels of Cd were six to twenty times higher in workers where it was detected. The highest recorded blood Cd levels were in the CRT and shredding workers who also showed higher Pb concentrations. Ceballos et al. suggests that the shredder is the cause, despite operating only three days per month.

In an interesting finding, Julander et al. (2014) reported concentration of In approximately twice as high in the serum of e-waste recycling workers in comparison to the office workers in the same building, ($p < 0.05$) (**Table 10**). In is used increasingly in electronics; mainly as indium-tin oxide in flat-screens. According to Julander et al. (2014), information on toxicity and carcinogenicity is scant and no reference concentration was available in the study of the Canadian population. The study cautions that with the increase in the number of flat-screens entering the after-use phase, that monitoring exposure in recycling workers in the context of their health is increasingly important.

Many workers in LIMICs, particularly those who are self-employed, run businesses from home and e-waste dismantlers are no exception according to the present study. Consequently, children may be exposed to potentially hazardous substances that are emitted during recovery of valuable materials whether the children are occupationally involved or otherwise. Five studies of children and one of pre-school children living in e-waste dismantling sites

measured blood concentrations of elements and compared them to non-exposed children (**Table 11**).

Levels of Pb were significantly higher in exposed children compared to the control groups in all six studies, providing a strong indication that e-waste recycling operations were the cause. Compared to the Canadian child population (Saravanabhavan et al., 2017) (detailed in **Section S.4.5**), blood Pb levels were more than double for pre-school children and between double and triple the level for all children in the other five studies. The United States Centers for Disease Control and Prevention stipulates a relative safety Pb concentration of 50 μg L^{-1} (Centers for Disease Control and Prevention, 2019) which it states is far below the 450 μg L^{-1} after which chelation therapy is required. Four of the studies reported values greater than 50 μg L^{-1}, one of which, Zhang et al. (2017), reported an average of more than double.

Table 11: Blood concentrations of selected elements compared for children living in e-waste recycling sites in China compared with reference groups (μg L^{-1}).

Ref.	Receptors	Activity context	n	Cd	Cr	Mn	Pb
Cai et al. (2019)	Preschool children	Living e-recycling area	358				48.8¥***
		Living non-e-recycling area	216				34.7¥
Liu et al. (2018)	Children	Living e-recycling site	146				49.4‡***
		Living non-e-recycling site	88				38.5‡
Zhang et al. (2017)	Children	Living e-recycling site	153	8.3‡			103.4‡***
		Living non-e-recycling site	141	1.79‡			23.9‡
Zheng et al. (2019)	Children	Living e-recycling site	104				72.3‡***
		Living non-e-recycling site	96				39.1‡
Zeng et al. (2017)	Children	Living e-recycling site	100	3.57‡			55.3‡***
		Living non-e-recycling site	106	0.57‡			5.8
Zeng et al. (2016)	Children	Living e-recycling site	300	0.576‡***	7.65†	28.18	62.4‡***
		Living non-e-recycling site	170	0.5‡	7.49†	20.09	47.5‡

The following reference values are applied as appropriate to the age group (**Section S.4.5**): ¥exceeds reference value for blood on non-occupationally exposed Canadian child citizens (three to five years of age); ‡exceeds reference value for blood on non-occupationally exposed Canadian child citizens (six to nineteen years of age); ***significantly (p<0.001) higher in e-waste recycling area compared to reference using independent sample t-test. or Mann-Whitney U test in the case of Zheng et al. (2019) or two sample T-Test in the case of Cai et al. (2019). Abbreviations: Number of samples (n).

Levels of blood Cd were tested in three of the six studies, and were found to be between two and thirteen times higher than the Canadian average (0.27 for six to nineteen year olds (Saravanabhavan et al., 2017)) in both the reference groups and the exposed groups. However, a significant difference between the exposed and control groups was only found by Zeng et al. (2016). Cd is widely used in electronic goods in batteries and printed circuit boards and is considered to be highly hazardous with reported effects including teratogenicity; endocrine and reproductive toxicities; nephrotoxicity, immunotoxicity, neurotoxicity, and cardiovascular disease (Zhang et al., 2017). Clearly, any indication that children are being exposed to elevated levels of Cd should be treated with concern.

3.1.5. Non-carcinogenic risk

To interpret concentrations of PTEs measured in human and environmental media, human health risk assessment models have been developed to evaluate the adverse health effects in receptors who may be occupationally or environmentally exposed via different pathways. Non-carcinogenic and carcinogenic effects are reported separately by calculating the average daily intake for a specific pathway/receptor combination; requiring the following parameters:

(1) Exposure frequency (day/year), exposure duration (year), the body weight and the average daily exposure time are common factors and are used for all exposure routes (ingestion, inhalation and dermal exposure);

(2) Ingestion rate and concentration of PTEs in soil, dust and/or food needed for the calculation of average daily intake via ingestion;

(3) Inhalation rate and concentration of PTEs in air needed for the calculation of 'acceptable daily intake' (ADI) via inhalation;

(4) Exposed skin surface area, skin adherence factor, dermal absorption factor and concentration of PTEs in soil, dust and/or air needed for the calculation of average daily intake via dermal exposure.

Recommended values for these exposure factors and further methodology description is provided by the United States Environmental Protection Agency (2001). Non-cancer risk effects are expressed using either a 'hazard index', a 'hazard quotient' or both. The hazard quotients are obtained by calculating the ratio of the average daily intake to the corresponding reference dose. If the ratio is greater than one, then the receptor is likely to experience adverse health effects; if it is greater than 10, the risk is considered high and the receptor may experience chronic health effects (Fang et al., 2013; Fujimori et al., 2012).

When exposure to more than one substance is being assessed, the hazard quotients can be summed to calculate a hazard index, thus indicating the risk from multiple hazards (Ohajinwa et al., 2019a). As with the hazard quotients, if the hazard index is greater than one, the potential for non-cancer risk is significant, if it is less than one, then the non-cancer risk is considered non-significant (Cao et al., 2020).

Two studies, Fang et al. (2013) and Huang et al. (2016) determined the non-carcinogenic risk through air inhalation for different receptors (**Table 12** and **Table 13**). Specifically, Huang et al. (2016) estimated the hazard index for non-occupationally exposed receptors (children and adults), while Fang et al. (2013) focused on occupationally exposed receptors. Instinctively, occupational exposure may be expected to result in higher risks compared to non-occupational environments. However the results showed higher non-cancer risk effects for children and residents living near e-recycling sites compared to the workers. This unexpected

discrepancy may be explained by differences in the two study approaches. Fang et al. (2013) measured the concentration of PTEs in ambient $PM_{2.5}$ air samples whereas Huang et al. (2016) measured whole air exposure from buildings very close to e-waste recycling workshops. Fang et al. (2013) also tested far fewer element types compared to Huang et al. (2016) with five and thirteen elements investigated respectively.

Table 12: Non-carcinogenic hazard index for several pathway/receptor combinations for receptors exposed to potentially toxic elements (PTEs) from e-waste dismantling activities.

Pathway	Media	Ref.	Geog.	Media description	Activity context	Receptor	n	Hazard index	PTEs of concern (HQ>1)	PTEs included
		Huang et al. (2016)	CHN	From three heights above the ground		Adults	165	2.7†	-	Pb, Cd, Cr, Cu, Fe, Mn, Ni, Zn, As, Sb, V, Co, Mo
					E-recycling site	Children	165	8†	Ni	
		Fang et al. (2013)	CHN		Mech. sorting area	Workers	2	<1	-	
Inhalation	Air			$PM_{2.5}$ air samples	Dis. area	Workers	2	<1	-	Pb, Cd, Cr, Cu, Ni
						Adults	41	1.9†	Pb	
				Topsoil (d=0-30 cm)	Dis. area	Children	41	14.9†	Pb, Cu, Fe	
						Adults	37	<1	-	
		Ackah (2019)	GHA	Subsoil (d=30-100 cm)	Dis. area	Children	37	4.9†	Pb	Pb, Cd, Cr, Cu, Fe, Zn, Ag, As, Se
						Adults	6	<1	-	
					E-recycling site (formal)	Children	6	<1	-	
		Fujimori et al. (2012)	PHL			Adults	7	<1	-	
				Topsoil (d=0-30 cm)	E-recycling site (informal)	Children	7	3.3†	-	Pb, Cd, Cu, Mn, Ni, Zn, Ag, Co
		Ohajinwa et al. (2019a)	NGA		Dis. Area *	Workers	n/a	<1	-	Pb, Cd, Cr, Cu, Fe, Mn, Ni, Zn, Ag, As, Sb, V, Co, Mo, Ga, Hg, Se
Ingestion	Soil			Topsoil (d=0-10 cm)	Repair area	Workers	n/a	<1	-	
						Adults	11	4.6†	Pb	
					E-recycling site (formal)	Children	11	37†	Pb	
		Fujimori et al. (2012)	PHL			Adults	6	1.4†	Pb	
				Surface dust	E-recycling site (informal)	Children	6	11†	Pb	Pb, Cd, Cu, Mn, Ni, Zn, As, Ag, Co, In
		Fang et al. (2013)	CHN		Mech. sorting area	Workers	n/a	7.28†	Pb	
				Floor dust	Dis. area	Workers	n/a	2.81†	Pb	Pb, Cd, Cr, Cu, Ni
		Zhang et al. (2019)	CHN			Children (2-7 yrs)	21	2.02†	Pb	
				Indoor and outdoor dust	E-recycling site	Children (8-19 yrs)	30	<1†	-	Pb, Cd, Cu, Zn
					Dis. area (control soil)**	Workers	n/a	<1	-	
				Floor dust (control soil)	Repair area (control soil)**	Workers	n/a	<1	-	
					Dis. area	Workers	n/a	<1	-	
				Roadside dust	Repair area	Workers	n/a	1†	-	
		Ohajinwa et al. (2019a)	NGA		Dis. area	Workers	n/a	<1	-	Pb, Cd, Cr, Cu, Fe, Mn, Ni, Zn, Ag, As, Sb, V, Co, Mo, Ga, Hg, Se
Ingestion	Dust			Direct dust from e-waste	Repair area	Workers	n/a	1.5†	-	

* From three different e-recycling sites; ** from two different e-recycling sites. Abbreviations: Geographical context (Geog.); number of samples (n); particulate matter < 2.5 µm ($PM_{2.5}$); hazard quotient (HQ); potentially toxic elements (PTEs); dismantling (dis.); mechanical (mech.); not available (n/a).

Table 13: Non-carcinogenic hazard index for several pathway/receptor combinations for receptors exposed to potentially toxic elements (PTEs) from e-waste dismantling activities.

Pathway	Media	Ref.	Geog.	Media description	Activity context	Receptor	n	Hazard index	PTEs of concern (HQ>1)	PTEs included
Dermal exposure	Soil	Ohajinwa et al. (2019a)	NGA	Topsoil (d=0-10 cm)	Dis. area *	Workers	n/a	2,300† – 4,000†	Cd, Cr, Cu, Fe, Mn, Ni, Zn, Ag, As, Sb, V, Hg	Pb, Cd, Cr, Cu, Fe, Mn, Ni, Zn, Ag, As, Sb, V, Co, Mo, Ga, Hg, Se
					Repair area	Workers	n/a	1,500†	Cr, Fe, Ni, Ag, As, Sb, V, Hg	
		Ackah (2019)	GHA	Topsoil (d=0-30 cm)	Dis. area	Adults	41	<1	-	Pb, Cd, Cr, Cu, Fe, Zn, Ag, As, Se
						Children	41	<1	-	
				Subsoil (d=30-100 cm)		Adults	37	<1	-	
					Dis. area	Children	37	<1	-	
Dermal exposure	Dust	Ohajinwa et al. (2019a)	NGA	Roadside dust	Dis. area (control soil) **	Workers	n/a	1,700†-5,500†	Cd, Cr, Cu, Fe, Mn, Ni, Zn, Ag, As, Sb, V, Hg	Pb, Cd, Cr, Cu, Fe, Mn, Ni, Zn, Ag, As, Sb, V, Co, Mo, Ga, Hg, Se
				Floor dust (control soil)	Repair area (control soil) *	Workers	n/a	1,400†-2,300†	Cr, Cu, Fe, Mn, Ni, Ag, Sb, V, Hg	
					Dis. area	Workers	n/a	2,000†	Cr, Cu, Fe, Mn, Ni, Zn, Ag, As, Sb, V, Hg	
				Roadside dust	Dis. area	Workers	n/a	5,400†	Cr, Fe, Mn, Ni, Ag, V, Hg, Co	
				Direct dust from electronics	Repair area	Workers	n/a	4,000†	Cr, Cu, Fe, Mn, Zn, As, Sb, V, Hg	
				Roadside dust	Repair area	Workers	n/a	1,800†	Cr, Fe, Mn, Ni, Ag, Sb, V, Hg	
Dermal exposure	Air	Fang et al. (2013)	CHN	$PM_{2.5}$ air samples	Mech. sorting area	Workers	n/a	<1	-	Pb, Cd, Cr, Cu, Ni
					Dis. area	Workers	n/a	<1	-	

* From three different e-recycling sites; ** From two different e-recycling sites. Abbreviations: Geographical context (Geog.); number of samples (n); particulate matter < 2.5 µm ($PM_{2.5}$); hazard quotient (HQ); potentially toxic elements (PTEs); dismantling (dis.); mechanical (mech.); not available (n/a).

Five studies calculated the hazard quotients as a result of soil and dust ingestion (**Table 12** and **Table 13**). The pathway for exposure to soil and dust is through deposition on food, hand-to-mouth activity and/or direct soil consumption, usually by children (Ohajinwa et al., 2019a). Two studies reported the non-cancer risk induced by soil ingestion for residents living in e-recycling sites indicating significant non-cancer risk for children and low risk for adults living near informal e-recycling areas (Ackah, 2019; Fujimori et al., 2012). For example, Ackah (2019) reported that hazard quotient of Pb was 7.5 times greater for children than for adults indicating the exposure to Pb via soil ingestion was the greatest contributor to non-cancer risk effect for children. In addition, Ohajinwa et al. (2019a) reported the hazard index through soil ingestion for workers in informal e-recycling as inconsiderable. These findings indicate that the specific exposure pathway (soil ingestion) is negligible for adults, including occupationally and non-occupationally receptors, but it is considerable for children due to their soil-pica behaviour and low body weight leading to higher soil ingestion rates (Fujimori et al., 2012; United States Environmental Protection Agency, 2017).

Zhang et al. (2019) reported a high hazard quotient from ingestion of Pb through dust for preschool children. These results are similar to Fujimori et al. (2012) who calculated the hazard index and quotients different PTEs caused by dust ingestion for adults and children living nearby formal and informal e-recycling operations. Interestingly, the hazard index was higher at formal e-recycling areas than informal sites attributed to the accumulation of PTEs in indoor surface dust of formal e-recycling sites, absence of any natural dilution effect such as rain and wind, and higher concentration of specific elements (for example, Pb, Cu, Ni and In) (Fujimori et al., 2012). In addition, Fang et al. (2013) and Ohajinwa et al. (2019a) determined the non-cancer risk for occupationally exposed receptors through dust ingestion in and around different e-recycling workshops related to dismantling. Fang et al. (2013) found high hazard index for workers in mechanical sorting and dismantling workshops caused by dust ingestion with Pb being the greatest contributor in both workshops (80% to 92% of hazard index). Ohajinwa et al. (2019a) found negligible non-cancer risks around the e-recycling workshops, except for workers in the repair area where the hazard index was higher than one due to dust ingestion directly from e-waste.

Substances present in soil and dust can adhere to exposed skin leading to dermal absorption and therefore dermal exposure to PTEs via soil and dust can occur (Ohajinwa et al., 2019a). However, limited information exists making it difficult to compare and assess the non-cancer

risk for this exposure pathway. Ackah (2019) found that dermal exposure induced negligible risk for children and adults living nearby e-recycling sites. However, Ohajinwa et al. (2019a) reported that dermal contact with soil and dust were the main pathways of exposure to PTEs for workers finding extremely high hazard index', which were attributed to the absence of PPE for the majority of workers (82% of total workers) (Ohajinwa et al., 2019a). The difference between Ackah (2019) and Ohajinwa et al. (2019a) for dermal exposure via soil is related to the selection of different dermal absorption factors, but still further scrutiny is required to obtain confidence about the contribution of dermal exposure to non-cancer risk effects.

Food consumption is also a pathway of exposure via ingestion, reported here as 'dietary intake' to distinguish from incidental dust and soil ingestion. The exposure to PTEs through dietary intake for different age group populations nearby e-recycling sites was investigated by Zhang et al. (2019) (**Table 14**), indicating that the consumption of vegetables and rice is likely to cause adverse non-cancer health effects. Hazard quotients higher than one found for Pb in vegetable consumption, whereas in rice consumption Cd and Cu had the greatest contribution to the hazard index (Zhang et al., 2019).

Table 14: Non-carcinogenic hazard index (HI) for different age group population living in China nearby e-waste dismantling activities exposed to potentially toxic elements (PTEs) via dietary intake reported by Zhang et al. (2019).

Food type	Receptor	n	Hazard index	PTEs of concern (HQ>1)	PTEs included
	Children (2 to 7 years)	21	2.51†	Pb	
	Children/ teenagers (8 to 19 years)	30	1.7†	-	
Vegetable	Adults (≥20)	88	1.47†	-	
	Children (2 to 7 years)	21	7.74†	Cd, Cu, Zn	
	Children/ teenagers (8 to 19 years)	30	5.94†	Cd, Cu	
Rice	Adults (≥20)	88	4.76†	Cd, Cu	
	Children (2 to 7 yr)	21	<1	-	
	Children/ teenagers (8 to 19 years)	30	<1	-	
Egg	Adults (≥20)	88	<1	-	
	Children (2 to 7 years)	21	<1	-	
	Children/ teenagers (8 to 19 years)	30	<1	-	
Drinking water	Adults (≥20)	88	<1	-	
	Children (2 to 7 years)	21	<1	-	
	Children/ teenagers (8 to 19 years)	30	<1	-	
Fish	Adults (≥20)	88	<1	-	
	Children (2 to 7 years)	21	<1	-	
	Children/ teenagers (8 to 19 years)	30	<1	-	Pb, Cd, Cu, Zn
Meat	Adults (≥20)	88	<1	-	

Abbreviations: Number of samples (n); hazard quotient (HQ); potentially toxic elements (PTEs).

The determination of hazard index reported by different authors for each exposure pathway was variable and appears inconsistent in places. It was beyond the resources available in the present study to compare the methods employed in detail, but there are clearly differences in the number and type of elements investigated and the absorption factors used, for example, Ackah (2019) and Ohajinwa et al. (2019a). Most of the researchers determined the total concentrations of PTEs in different sources (for example, soil, dust, air and food) for the calculation of non-cancer risk indices (hazard index and hazard quotients) leading to a potential overestimation of human health risk effects. An alternative approach would have been to determine the concentration of bio-accessible PTEs which has been reported to provide a more accurate representation of risk (Cao et al., 2020; Oguri et al., 2018). Cao et al. (2020) reported that the exposure risk can be assessed with higher accuracy by using bio-accessible concentrations of PTEs and bio-accessibility-corrected human health risk assessment was recommended.

3.1.6. Carcinogenic risk

The carcinogenic risk is the incremental probability of a receptor, exposed to carcinogenic hazards, to develop cancer over a lifetime. The carcinogenic risk is determined by average daily intake and the cancer slope factor, for which indicative values are provided by US Environmental Protection Agency (United States Environmental Protection Agency, 2019a). Tolerable cancer risk levels range between 10^{-6} and 10^{-4} depending on the situation and circumstances of exposure (NHMRC, 2010; United States Environmental Protection Agency, 1991). Levels less than 10^{-6} indicate that cancer risk is negligible, while values greater than 10^{-4} are unacceptable (NHMRC, 2010). As with the non-cancer risk assessments, cancer risks resulting from PTE exposure for specific pathway/receptor combinations in e-waste dismantling activities were also investigated (**Table 15**).

Table 15: Cancer risk (hazard index) from potentially toxic element (PTE) exposure reported for several pathway-receptor scenarios in e-waste dismantling zones.

Pathway	Media	Ref.	Geog.	Media description	Activity context	Receptor	n	Hazard index	PTEs included
						Adults	165	1.3E-03†	
						Children	165	3.9E-03†	Cr, Co, Ni, As, Cd
						Adults	165	5.4E-05	
		Huang et al. (2016)	CHN	From three heights above the ground	E-recycling site	Children	165	1.6E-04†	Co, Ni, As, Cd
							2	3.5E-04†	Cr
							2	9.4E-06	Ni
							2	1.5E-06	Cd
				$PM_{2.5}$ air samples	Mech. sorting area		2	5.6E-06	Pb
							2	1.4E-04†	Cr
							2	6.6E-06	Ni
							2	2.0E-06	Cd
				$PM_{2.5}$ air samples	Dis. area	Workers	2	9.9E-06	Pb
							2	1.0E-04†	Cr
							2	3.7E-06	Ni
							2	3.3E-07	Cd
Inhalation	Air	Fang et al. (2013)	CHN	$PM_{2.5}$ air samples	Southeast of workshops	Residents	2	3.3E-07	Pb
						Children	41	2.6E-04†	
				Topsoil (d=0-30cm)		Adults	41	3.6E-05	
						Children	37	8.0E-06	
		Ackah (2019)	GHA	Subsoil (d=30-100cm)	Dis. area	Adults	37	1.1E-06	As
		Ohajinwa et al. (2019a)	NGA		Dis. area*		n/a	2.3E-05 - 6.8E-05	
Ingestion	Soil			Topsoil (d=0-10 cm)	Repair area	Workers	n/a	1.2E-05	Cr, Co, Ni, As, Cd, Hg, Pb
					Dis. area**		n/a	1.9E-05 - 8.9E-05	
				Floor dust (control soil)	Repair area*		n/a	1.1E-05 - 8.7E-05	
					Dis. area		n/a	2.1E-05	
				Roadside dust	Repair area		n/a	6.0E-06	
		Ohajinwa et al. (2019a)			Dis. area		n/a	3.4E-06	
Ingestion	Dust		NGA	Direct dust from e-waste	Repair area	Workers	n/a	3.1E-05	Cr, Co, Ni, As, Cd, Hg, Pb
		Ohajinwa et al. (2019a)			Dis. area**		n/a	2.8E-03† - 21E-03†	
Dermal exposure	Soil		NGA	Topsoil (d=0-10 cm)	Repair area	Workers	n/a	1.2E-03†	Cr, Co, Ni, As, Cd, Hg, Pb
				Floor dust (control soil)	Dis. area**	Workers	n/a	3.1E-03† - 35E-03†	Cr, Co, Ni, As, Cd, Hg, Pb

Pathway	Media	Ref.	Geog.	Media description	Activity context	Receptor	n	Hazard index	PTEs included
Dermal exposure	Dust	Ohajinwa et al. (2019a)	NGA		Repair area*		n/a	7E-04 - 52E-04†	
				Roadside dust	Dis. area		n/a	5.2E-03†	
					Repair area		n/a	4.1E-04†	
				Direct dust from e-waste	Dis. area		n/a	3.9E-04†	
					Repair area		n/a	4.2E-03†	

† Exceeds target value of 1.0E-04 indicating unacceptable risk. Abbreviations: Geographical context (Geog.); number of samples (n); particulate matter < 2.5 µm (PM2.5); potentially toxic elements (PTEs); dismantling (dis.); mechanical (mech.); not available (n/a).

The cancer risk induced by air inhalation for non-occupationally exposed receptors (adults and children) was determined by Huang et al. (2016) and for occupationally exposed receptors by Fang et al. (2013). Huang et al. (2016) found high cancer risk for both children and adults living nearby e-waste dismantling activities with Cd being the main contributor. The cancer risk calculated by Huang et al. (2016) approaches almost the threshold value if Cd is excluded, indicating that Cd is the main reason for high cancer risk via air inhalation for both receptors. In addition, Fang et al. (2013) ranked the PTE's contribution to lifetime cancer risk in the following order Cr > Ni > Pb > Cd, indicating that Cr poses the highest cancer risk on workers.

For the ingestion of PTEs via soil pathway, the cancer risk for different receptors was investigated by Ackah (2019) and Ohajinwa et al. (2019a). In Nigerian e-waste dismantling sites, Ohajinwa et al. reported no cancer risk through ingestion of soil. Ackah also reported no risk for adults; however, the risk of cancer to children from topsoil was greater than the target value, indicating an unacceptable level of risk.

Limited information can be found for the cancer risk as a result of dermal exposure to contaminated soil or dust by PTEs. Only Ohajinwa et al. (2019a) reported considerably high values for both sources (dust and soil), indicating the need for further investigation on human health risk of dermal exposure to PTEs in e-recycling areas.

3.2. Flame retardants

3.2.1. Context

Plastics used in electrical and electronic goods are often modified with additives to inhibit combustion in the event of malfunction. These additives are known collectively as 'flame retardants' and comprise several groups of substances, including brominated flame retardants (BFRs) and chlorinated paraffins (CP). Both groups of substances include multiple formulations, some of which are potentially hazardous (National Research Council (US) Subcommittee on Flame-Retardant Chemicals, 2000).

The BFR group comprises 209 congeners divided into three formulas: Penta-BDE, Octa-brominated diphenyl ether (BDE) and Deca-BDE (McGrath et al., 2017). An increasing body of evidence exists which implicates multiple BFR congeners in serious health effects including cancer, diabetes, reproductive health, neurobehavioral and developmental

disorders, and alteration in thyroid function (Hong-Gang et al., 2016; Kim et al., 2014; McGrath et al., 2017). Penta-BDE, Octa-BDE and Deca-BDE formulations are all listed as persistent organic pollutants (POPs) under the Stockholm Convention (Tang et al., 2014; UN Environment, 2017) and production of both Penta-BDE and Octa-BDEs is banned in the US (Venkatesan and Halden, 2014) and Europe, despite continued production in China (Tang et al., 2014).

More than 200 formulations of CPs exist which can be broadly classified by the length of the polymer as follows: short-chain CPs (SCCPs, C_{10-13}), medium chain CPs (MCCPs, C_{14-}17) and long-chain (LCCPs, $C_{>17}$) (National Research Council (US) Subcommittee on Flame-Retardant Chemicals, 2000). CPs have generally low acute toxicity and a recent health risk assessment carried out by the EFSA Panel on Contaminants in the Food Chain (CONTAM), was inconclusive (Efsa Panel on Contaminants in the Food Chain et al., 2020). Nonetheless, CPs have high bioaccumulation potential, particularly for lower chlorinated species, and SCCPs are carcinogenic to rats and some toxicity has been observed in aquatic organisms (Iozza et al., 2008). SCCPs have also been listed as POPs under the Stockholm Convention (Secretariat of the Stockholm Convention, 2017), as possibly carcinogenic to humans by the International Agency for Research on Cancer (IARC) and as hazardous under the European Water framework Directive.

3.2.2. Air

Environmental release rates of BFRs and CPs are only partly understood (Webster et al., 2009) and in this research, incomplete information was identified to evidence emissions from sorting, dismantling and physical processing of e-waste. Many authors also report results which are potentially confounded, especially in LIMICs where open burning may take place in proximity to non-thermal processing. The four studies of flame retardant concentrations in air presented in **Table 16** all reported data from facilities where, according to the text, open burning did not appear to be taking place. One of these, Muenhor et al. (2010) reported concentrations of PBDEs in the air at five e-waste storage facilities in Thailand. Referring to **Table S 14** in **Section S.4.6** (supplementary information) many of the concentrations reported by Muenhor et al. (2010) were not particularly high in comparison to mean concentrations reported in outdoor urban air in the USA (0.052, 0.1 ng m^{-3}), Italy (0.106 ng m^{-3}) and China (0.045 ng m^{-3}). However, several were higher than those reported in Spain (0.035, 0.018 ng m^{-3}), the UK (0.018 ng m^{-3}), Sweden (0.0063 ng m^{-3}), Greece (0.026, 0.015

ng m^{-3}), Australia (0.0093 ng m^{-3}), and China (0.019, 0.025, 0.045 ng m^{-3}). Notably, concentrations in the indoor air at Facility 5 were approximately an order of magnitude higher than many of three other observed concentrations, indicating the possibility that migration and passive release is a potentially significant source of BFR emissions. Nonetheless, the concentrations were still an order of magnitude lower than computer laboratories in California investigated by Cahill et al. (2007).

Table 16: Flame retardant concentrations in air at e-waste storage and processing facilities (ng m^{-3}).

Author	Geog.	Activity context	Sample	n	ΣPBDE	a-DP	s-DP	ΣCP	ΣotherBFR
			Indoor	1	0.046				
			Outdoor upwind	1	0.024				
			F1 Outdoor downwind	1	0.075				
			Indoor	1	0.052				
			Outdoor upwind	1	0.012				
			F2 Outdoor downwind	1	0.008				
			Indoor	1	0.14				
			Outdoor upwind	1	0.061				
			F3 Outdoor downwind	1	0.11				
			Indoor	1	0.35				
			Outdoor upwind	1	0.15				
			F4 Outdoor downwind	1	0.057				
			Indoor	1	0.046				
			Outdoor upwind	1	0.033				
			F5 Outdoor downwind	1	0.023				
			Mean		0.079				
Muenhor et al. (2010)	THA	E-waste storage	Median		0.052				
		Recycling (control)*	Facility	15	88	0.71	0.36	1.2	0.2
			Small facility	22	320	3.3	3.1	6.5	98
			Medium facility	30	810	10	6.3	16	85
Gravel et al. (2019b)	CAN	E-waste recycling	Large facility	36	6,600	27	14	41	150
			Respirable dust	3	6.17				
			Inhalable dust	4	214.27				
Julander et al. (2005)	SWE	E-waste dismantling	Total dust	4	33.35				
			Computers switched off	2	1.4				
		Computer lab. (control)	Computers switched on	6	1.8				
			Shredding [a]	4	650				
Cahill et al. (2007)	USA	E-waste dismantling	No shredding [a]	2	93				

*Facility reported to recycle commercially sourced glass and a small amount of aluminium and cardboard; [a] samples were taken at the same plant on days when the shredder was active and inactive. Abbreviations: facility (F); number of samples (n); Geographical context (Geog.); polybrominated diphenyl ethers (PBDE); chlorinated paraffins (CP); Dechlorane Plus (DP); brominated flame retardants (BFR).

As with PTE emissions discussed in **Section 3.1**, concentrations of flame retardants measured in air at HIC e-waste physical processing plants provide a useful benchmark of the potential

emissions from non-thermal processing because open burning and de-soldering activities are highly unlikely to be taking place. The studies by Gravel et al. (2019b), Julander et al. (2005) and Cahill et al. (2007) provide such a context as they were carried out in HICs, showing a range of values at each of the plants being studied. Air concentrations at the control site determined by Gravel et al. (2019b) were comparable with many of the concentrations measured by Muenhor et al. (2010), however the concentrations in the medium and large plants were considerably higher, with the large facility measuring two orders of magnitude greater than the control. Of the 6,600 ng m^{-3} mean concentration detected at the large plant, 6,500 ng m^{-3} (range 5,300–8,400) was the BDE 209 congener (specific congeners not shown).

The study of a US e-waste dismantling plant by Cahill et al. (2007) showed similar concentrations of BFRs when the shredding equipment was switched on compared to the medium plant investigated by Gravel et al. (2019b). When the shredder was not operational, the BFR concentrations decreased by approximately an order of magnitude. This study provides compelling evidence to support the theory that BFRs are emitted during shredding activities.

Gravel et al. (2019b) also determined concentrations of two Dechlorane plus™ stereoisomers, anti-DP and syn-DP. The concentrations are hard to contextualise because although dechloranes have been in production in increasingly large quantities since the 1960s, their detection in the environment wasn't reported until 2006 (Sverko et al., 2011). Since then, dechloranes have been identified in multiple global locations, and their detection and potential hazardousness are an ongoing area of research. Due to their low water solubility and high octanol-water partition coefficient, in 2019 a proposal was submitted to the Stockholm Convention to have dechloranes added to Annexes A, B and/or C (Secretariat of the Stockholm Convention on Persistent Organic Pollutants, 2019). While acknowledging the persistence in the environment of dechloranes, the intersessional working group of the Stockholm Convention indicates acute toxicity is unlikely, although longer term studies have not been carried out (Secretariat of the Stockholm Convention, 2020).

3.2.3. Dust and soil

Studies of flame-retardants in soils and dusts in and around e-waste physical dismantling and processing facilities are another indication of emissions, evidenced in four studies, Chen et al.

(2018), Ohajinwa et al. (2019a), Muenhor et al. (2010) and Ma et al. (2009) (**Table 17**) with three further studies, Wang et al. (2013), Harrad et al. (2008) and Iwegbue et al. (2019) shown for comparison. Chen et al. (2018) measured concentrations of CPs in e-recycling workshop dust in China, finding concentrations of SCCPs and MCCPs ten times higher than in residential areas 2 km away and approximately 76 (SCCPs) and 80 (MCCPs) times higher than the reference homes 30 to 40 km away. All the concentrations of SCCPs were considerably higher than those observed by Wang et al. (2013) whose highest measurement at the roadside in China was two orders of magnitude lower than the concentrations identified in reference homes by Chen et al. (2018).

Table 17: Flame retardant concentrations observed in dust soil in and around e-waste dismantling facilities.

Author	Geog.	Activity context	Sample	n	Unit	ΣPBDE		ΣSCCP	ΣMCCP
						Mean	Median		
		E-recycling workshop		41				3,760	13,000
		Residents <2 km		30				370	1,130
		Nearby streets		10				359	567
Chen et al. (2018)	CHN	Ref. homes (30-40 km)	Dust	15	μg g^{-1}			49	162
		Farmland		18				0.0012-0.21	
		Roadside		6				0.031-0.42	
Wang et al. (2013)	CHN	Woodland	Soil	3	μg g^{-1}			0.00042-0.014	
		Dismantling sites				1.579			
		Repair sites	Soil*			0.00319			
		Dismantling sites				3.134			
		Repair sites	Floor dust (control soil)			0.452			
	NGA, Lagos	Roadside at dismantling sites	Dust			1.06			
		Dismantling sites	Soil *			0.018			
		Repair sites	Floor dust (control soil)			8.22			
		From electronics				3.594			
	NGA, Ibadan	From electronics	Dust (direct)	Soil (n=16)		1.685			
		Repair sites	Floor dust	Floor dust (n=29) Roadside dust (n=5)		0.55315			
Ohajinwa et al. (2019a)	NGA, Aba	Roadside	Dust (control RS)	Direct (n=6)	ng g^{-1}	0.1114			
Muenhor et al. (2010)	THA	E-waste storage facilities	Dust	25	ng g^{-1}	43,000$^{\mp\sharp}$			
			Dust	5		1,910			
			Soil	10		30,700$^{\mp\sharp}$			
			Leaf	6		30.6			
Ma et al. (2009)	CHN	E-waste dismantling shredding facility	Urban soil (Ref.)	3	ng g^{-1}	0.30			
		Homes		30		260,000$^{\mp\sharp}$	8,500$^{\sharp}$		
		Offices		18		31,000$^{\mp\sharp}$	7,400$^{\sharp}$		
Harrad et al. (2008)	GBR	Cars	Dust	20	ng g^{-1}	340,000$^{\mp\sharp}$	57,000$^{\mp\sharp}$		
		Computer/photocopier/printer workshops		10		457	366		
		TV/radio/stereo/video player workshops		20		1,112	958		
Iwegbue et al. (2019)	NGA	Telephone/tablet workshops	Dust	10	ng g^{-1}	622	704		

* d=0-10 cm; dust (direct) means dust collected directly from the circuit-boards inside electronic equipment; soil and dust concentrations benchmarked against USEPA screening levels (detailed in **Section S.4.7**) as follows: † exceeds USEPA carcinogenic screening level (TR=1E-06) for BDE-209 in industrial soils; ‡ exceeds USEPA non-carcinogenic screening level (TR=1E-06) for BDE-209 in industrial soils; $^{\mp}$ exceeds USEPA non-carcinogenic screening level (TR=1E-06) for BDE-153 in industrial soils; $^{\sharp}$ exceeds USEPA non-carcinogenic screening level (THI=0.1) for BDE-47 and BDE-99 in industrial soils. Abbreviations: Polybrominated diphenyl ether (PBDE); SCCP (short chain paraffins); MCCP (medium chain paraffins); number of samples (n); Geographical context (Geog.); reference (Ref.).

The PBDE concentrations reported by Ohajinwa et al. (2019a) were extremely low in comparison to all other authors other than the reference urban soil samples analysed by Ma et al. (2009). Even the highest concentrations identified by Ma et al. (2009) and Muenhor et al. (2010) were lower than dusts analysed by Harrad et al. (2008) in UK offices, and lower by an order of magnitude compared to UK cars and homes. While initially surprising, it is important to note that the likely source of the PBDEs in the study by Harrad et al. (2008) is soft furnishings, in which plastic fibres have a much higher surface area compared to electrical items. While it is beyond the scope to consider these mechanisms in more detail, it is suggested that dust accumulating on the soft furnishing fibres act as a conduit for PBDE migration. Some evidence for this is also indicated by Ohajinwa et al. (2019a) who found some of the highest concentrations of PBDEs on dust sampled from the surface of electronic circuit boards, indicating that migration into dust is a significant pathway through which PBDEs migrate from plastic electrical components.

Whereas many papers reviewed here mentioned non-thermal emissions of BFRs, discussion of the mechanism of release was brief or non-existent, focusing on open burning, which is thought to be one of the main mechanisms of BFR emissions (Hong-Gang et al., 2016). Non-thermal emissions of BFRs or CPs require their migration from the spaces between the host polymer chains to the surface of the plastic (Stubbings and Harrad, 2014), which may be hastened through the increased surface area created through comminution, the latter of which was suggested as a potential source of BFR emissions at rural e-waste processing sites in China by Tian et al. (2011). Other BFRs, such as TBBPAs, are less likely to be released through migration as they are principally used as a reactant in epoxies, phenolic resins and polycarbonate.

3.2.4. Non-carcinogenic risk

Only one study in Nigeria by Ohajinwa et al. (2019a) calculated hazard quotients for non-carcinogenic risk from exposure to BDE congeners from e-waste dismantling and repair activities (**Table 18**). Direct dust from electronics handled in the repair are resulted in the greatest risk to workers, inside e-waste dismantling and repair areas through dermal contact. One site, Ibadan, returned the highest hazard index, as well as Aba, whereas the Lagos site showed an index of below two in all exposure pathway categories. The ingestion pathway was only calculated to pose a risk from exposure to dust that had been in direct contact with circuitry at the Ibadan dismantling workshop, whereas the pathway did not result in hazard index of greater than one for any other context.

Table 18: Non-carcinogenic hazard indices reported by Ohajinwa et al. (2019a) for polybrominated diphenyl ether (PBDE) exposure to workers in Nigerian e-waste dismantling and repair workshops.

					Hazard quotients				
Pathway	**Sample**	**Geog.**	**Sample source location**		**PBDE-47**	**PBDE-99**	**PBDE-153**	**PBDE-209**	**Hazard index**
				Dismantling area	340††	810††	88†	2,300††	3,700†
		NGA, Ibadan	Direct dust from e-waste	Repair area	78††	140††	33†	110†	360†
				Dismantling area	<1	<1	<1	<1	<1
		NGA, Lagos	Floor dust	Repair area	<1	<1	<1	<1	<1
		NGA, Ibadan		Repair area (control soil)	22††	57††	20††	610††	710†
		NGA, Aba	Floor dust (control soil)	Repair area (control soil)	7.6†	14††	4.3†	41†	66†
		NGA, Lagos		Dismantling area	<1	<1	<1	<1	<1
	Dust	NGA, Aba	Roadside dust	Repair area	21†	31†	21†	5.2†	78†
				Dismantling area	<1	<1	<1	<1	<1
		NGA, Lagos		Repair area	<1	<1	<1	<1	<1
Dermal exposure	Soil	NGA, Ibadan	Topsoil (d=0-10 cm)	Dismantling area	1.1†	2.2†	1.3†	<1	5.6†
				Dismantling area	<1	<1	<1	1.74†	2.64†
		NGA, Ibadan	Direct dust from e-waste	Repair area	<1	<1	<1	<1	<1
				Dismantling area	<1	<1	<1	<1	<1
		NGA, Lagos			<1	<1	<1	<1	<1
		NGA, Ibadan			<1	<1	<1	<1	<1
		NGA, Aba	Floor dust (control soil)	Repair area	<1	<1	<1	<1	<1
		NGA, Lagos		Dismantling area	<1	<1	<1	<1	<1
	Dust	NGA, Aba	Roadside dust	Repair area	<1	<1	<1	<1	<1
				Dismantling area	<1	<1	<1	<1	<1
		NGA, Lagos		Repair area	<1	<1	<1	<1	<1
Ingestion	Soil	NGA, Ibadan	Topsoil (d=0-10 cm)	Dismantling area	<1	<1	<1	<1	<1

† Exceeds target value of one, indicating unacceptable risk; †† exceeds target value of 10, indicating very high risk. Abbreviations: polybrominated diphenyl ether (PBDE); number of samples (n); Geographical context (Geog.).

3.2.5. Carcinogenic risk

As with the non-carcinogenic risks, the carcinogenic risks from e-waste dismantling activities were highest from exposure to dust on circuit boards at the e-waste dismantling workshop in Nigeria (**Table 18**).

Table 19: Cancer risk for several pathway/receptor combinations induced by e-waste dismantling activities (specific congener BDE-209); after Ohajinwa et al. (2019a).

Pathway	Sample	Geog.	Media	Activity context	Receptor	Cancer risk (CR)
				Dismantling area		5.6E-07
		NGA, Lagos		Repair area		3.2E-10
	Soil	NGA, Ibadan	Topsoil (d=0-10 cm)	Dismantling area	Workers	4.8E-06
				Dismantling area		1.1E-06
		NGA, Lagos	Floor dust (control soil)	Repair area		1.6E-07
		NGA, Lagos	Roadside dust	Dismantling area		3.6E-07
		NGA, Ibadan	Floor dust (control soil)	Repair area		3.0E-03†
				Dismantling area		1.2E-02†
		NGA, Ibadan	Direct dust from e-waste	Repair area		5.5E-04†
Dermal exposure			Floor dust (control soil)			2.1E-04†
	Dust	Aba, Nigeria	Roadside dust	Repair area	Workers	2.6E-05
	Soil	NGA, Ibadan	Topsoil (d=0-10 cm)	Dismantling area	Workers	3.5E-09
				Dismantling area		8.1E-10
		NGA, Lagos	Floor dust (control soil)	Repair area		1.2E-10
		NGA, Lagos	Roadside dust	Dismantling area		2.6E-10
		NGA, Ibadan	Floor dust (control soil)	Repair area		2.2E-06
				Dismantling area		8.5E-06
		NGA, Ibadan	Direct dust from e-waste	Repair area		3.9E-07
			Floor dust (control soil)			1.4E-07
Ingestion	Dust	Aba, Nigeria	Roadside dust	Repair area	Workers	1.9E-08

† Exceeds target value of 1.0E-04 for BDE-209 or 1.0E-06 for polycyclic aromatic hydrocarbons (PAH) indicating unacceptable risk. Abbreviations: Geographical context (Geog.).

3.3. Risk characterisation for sorting, dismantling and physical processing of e-waste

The qualitative risk assessment scores for sorting, dismantling and physical processing of e-waste are shown in **Table 20**. Informal waste workers were scored with very high risk for dermal contact with PTEs and medium to high risk for dermal contact with BFRs and inhalation of PTEs during physical processing of e-waste. Formal workers also scored medium to high risk for inhalation of PTEs while working in proximity to mechanical processing as despite better standards of protective equipment and safety procedures in general, it is suggested that the more highly mechanised systems result in considerably greater emissions compared to the less mechanised informal sector operations. However, the carcinogenic risk of exposure to PTEs has been reported to exceed the acceptable level (10^{-6}-

10^{-4}) for informal workers, as a result of limited accessibility to protective equipment risk of ingestion, high level of dermal contact and inhalation. Children living on e-waste sites are also at high risk from ingestion of PTEs because they eat more soil than adults and at medium to high risk from PTEs uptake in food. Specifically, the levels of Pb and Cd in blood of children living nearby e-waste dismantling activities were found to be concededly higher compared to non-exposed children. The risk for population nearby e-waste recycling activities is medium to high due to elevated levels of PTEs identified in dust and soil of surrounding areas, but still the risk is higher for children compared to adults.

Apart from the very high risk to informal workers identified, the risk from BFRs to the general population was low for children and adults as the detection of these substances in environmental media was not sufficient to pose a significant threat.

Table 20: Risk characterisation summary for sorting, dismantling and physical processing of e-waste (non-thermal).

Haz.	Pathway	Receptor	Geog.	Evidence and justification for risk assessment	Uncertainty (aleatoric and epistemic)	Receptor vulnerability	L	S	R	Global receptor context
PTE	Atmosphere/ Inhalation	E-waste workers (formal)	CHN, SWE, USA, GHA	• Clear evidence (Ceballos et al., 2017; Julander et al., 2014) of emissions to air of several PTEs through physical processing in HICs, potentially greater than in LIMICs due to more intensive mechanised processing. • Dismantling workers significantly more exposed compared to office workers for most elements (Ceballos et al., 2017). • Emissions of Pb and Cd exceeded guidelines (**Section S.4.1)** in two examples relating to shredding and CRT processing (Ceballos et al., 2017). • Significantly elevated blood concentrations of Cr and Pb for those engaged in HIC physical processing activities (Julander et al., 2014) and exceeding reference value for non-exposed Canadian citizens (Ceballos et al., 2017). • Elevated Pb in dismantlers urine significantly higher than office workers (Julander et al., 2014).	• Limited information on specific processing activities creates challenge to establish causality from particular processes.	• Use of PPE sometimes optional and not always used (Ceballos et al., 2017; Julander et al., 2014).	3	4	**12**	HIC
		E-waste workers (informal)		• Evidence for informal sector indicates concentrations low in areas where physical processing takes place (Huang et al., 2016; Tang et al., 2015; Zeng et al., 2016), however evidence of elevated blood Pb in one study (Amankwaa et al., 2017). • Non-carcinogenic hazard index indicates low risk to informal waste workers in two scenarios studied (Fang et al., 2013). • Carcinogenic risk indicates slightly above 1.3E-03 in one case (Fang et al., 2013).	• Results may be confounded with open burning and smelting that may take place in the same location.	• Informal workers operate without respiratory protective equipment.	3	4	**12**	LIMIC
		Population		• Non-carcinogenic risk to adults and children high in two examples (Huang et al., 2016) . • Carcinogenic risk to population (Huang et al., 2016) indicated in four examples. • Significantly higher Cd and Pb for residents living near to e-waste dismantling (Zhang et al., 2019).		• Adults and children have no choice to avoid exposure if they live around e-waste dismantling activities.	2	4	**8**	LIMIC
		Children		• Children living in e-waste areas have significantly higher Pb and Cd in blood compared to non-exposed in multiple studies (Cai et al., 2019; Liu et al., 2018; Zeng et al., 2017; Zeng et al., 2016; Zhang et al., 2017; Zheng et al., 2019), however inhalation is not considered a very prevalent source of exposure to children.	• Blood levels confounded with other exposure pathways, particularly ingestion.	• Children have no choice to avoid exposure if they live around e-waste dismantling activities.	3	4	**12**	LIMIC
PTE	Ingestion of dust, soil and water	E-waste workers (informal)	PHL, GHA, NGA, THA, IND, GRC, CHN	• Significant non carcinogenic risk identified in one case, specifically dust but low in all others (Ohajinwa et al., 2019a).	• Results may be confounded with open burning and smelting that may take place in the same location.	• Low hygiene means informal workers at risk of ingesting, particularly dust that they come into contact with.	1	4	**4**	LIMIC
		Population		• Elevated levels of PTEs identified in multiple dust and soil samples close to e-waste dismantling activities (Ackah, 2019; Chakraborty et al., 2019; Damrongsiri et al.,		• Adults and children have no choice to avoid exposure if they live	3	4	**12**	LIMIC

Haz.	Pathway	Receptor	Geog.	Evidence and justification for risk assessment	Uncertainty (aleatoric and epistemic)	Receptor vulnerability	L	S	R	Global receptor context
				2016; Fang et al., 2013; Fujimori et al., 2012; Isimekhai et al., 2017; Ohajinwa et al., 2018; Tang et al., 2015; Tzoraki et al., 2019) in LIMICs and one in an HIC. • Cd, Pb, Cr and Mn with multiple examples of exceeding USEPA guidelines. • Levels of PTEs low in few samples of fresh water and very low in tap water analysed (Tang et al., 2015; Zhang et al., 2019) in LIMICs but elevated levels of As, Cd, Cr, Pb in one HIC context (Tzoraki et al., 2019). • Significantly higher Cd and Pb for residents living near to e-waste dismantling (Zhang et al., 2019). • Children living in e-waste areas have significantly higher Pb and Cd in blood compared to non-exposed in multiple studies (Cai et al., 2019; Liu et al., 2018; Zeng et al., 2017; Zeng et al., 2016; Zhang et al., 2017; Zheng et al., 2019). • Significant non-carcinogenic risk through ingestion of dust and soil to adults and children in several cases (Ackah, 2019; Fujimori et al., 2012; Zhang et al., 2019). • Carcinogenic risk only identified in one case for children (Ackah, 2019).		around e-waste dismantling activities.				
		Children		• Children living in e-waste areas have significantly higher Pb and Cd in blood compared to non-exposed in multiple studies (Cai et al., 2019; Liu et al., 2018; Zeng et al., 2017; Zeng et al., 2016; Zhang et al., 2017; Zheng et al., 2019). • High risk to children living in and near e-waste dismantling activities (Ackah, 2019; Fujimori et al., 2012). • Carcinogenic risk to children identified in one case (Ackah, 2019).		• Children ingest soil and have no choice to avoid exposure if they live around e-waste dismantling activities.	4	4	16	LIMIC
		Population		• Some evidence (Tang et al., 2015; Zhang et al., 2019) provided for concentrations in food near e-waste physical processing, with three examples showing slightly higher Pb concentration than EU regulatory limits (European Union, 2006; European Union, 2015), however evidence is not strong enough to correlate with the e-waste processing activities. • High non-carcinogenic risk through dietary intake highlighted in one study particularly through dust settling on vegetables and rice (Zhang et al., 2019).	• Limited information indicates that many PTEs are not easily up-taken by rice plants (Wu et al., 2019), however further sources were not reviewed. • Assessment is based on a single estimate of non-carcinogenic risk in China (Zhang et al., 2019).	• Local population may have limited ability to choose whether to eat food from local area or not.	• 3	4	12	
	Uptake in food	Children	CHN	• Non-carcinogenic risk through dietary intake highlighted in one study, particularly through dust settling on vegetables and rice (Zhang et al., 2019), indicated slightly higher for children.			• 3	4	12	LIMIC
PTE	Dermal contact with soil and dust	Population	PHL, GHA, NGA, THA, IND, GRC, CHN	• Multiple examples of PTE concentrations in soils and dusts (Ackah, 2019; Chakraborty et al., 2019; Damrongsiri et al., 2016; Fang et al., 2013; Fujimori et al., 2012; Isimekhai et al., 2017; Ohajinwa et al., 2018; Tang et al., 2015; Tzoraki et al., 2019) in local environment. • Non-carcinogenic risk low according to one study (Ackah, 2019) because contact with dusts limited in comparison to occupational exposure.	• Results may be confounded with open burning and smelting that may take place in the same location.	• Adults and children have no choice to avoid exposure if they live around e-waste dismantling activities.	1	4	4	LIMIC

Haz.	Pathway	Receptor	Geog.	Evidence and justification for risk assessment	Uncertainty (aleatoric and epistemic)	Receptor vulnerability	L	S	R	Global receptor context
		E-waste workers (informal)		• Multiple examples of PTE concentrations in soils and dusts (Ackah, 2019; Chakraborty et al., 2019; Damrongsiri et al., 2016; Fang et al., 2013; Fujimori et al., 2012; Isimekhai et al., 2017; Ohajinwa et al., 2018; Tang et al., 2015; Tzoraki et al., 2019), particularly dust on electronic components and circuitry that are frequently handled by informal waste workers (Ohajinwa et al., 2018). • Non-carcinogenic and carcinogenic risk considered extremely high for both soils and dusts (Ohajinwa et al., 2019a). • One study (Fang et al., 2013) reports low risk through dermal contact with PM2.5 deposited from atmosphere, though this may be a minor source in comparison to direct electronic dusts.		• Informal workers operate without dermal protective equipment.	4	4	**16**	LIMIC
		E-waste workers (formal)		• Evidence for release into atmosphere as a consequence of dismantling and physical processing in HICs is robust, including an inferred link between increased physical processing, such as shredding and great emissions (Cahill et al., 2007; Gravel et al., 2019b; Julander et al., 2005). • Carcinogenic and non-carcinogenic risk not investigated through this pathway.	• Results may be confounded with open burning and smelting that may take place in the same location. • Limited occupational exposure data in LIMICs	• Use of PPE sometimes optional and not always used (Ceballos et al., 2017; Julander et al., 2014).	1	4	**4**	HIC
BFR	Atmosphere/ Inhalation	E-waste workers (informal)	CAN, THA, SWE, USA			• Informal workers operate without respiratory protective equipment.	1	4	**4**	LIMIC
		Population		• Though concentrations have been determined in soils and dusts in surrounding e-waste dismantling areas (Chen et al., 2018; Ohajinwa et al., 2019b; Wang et al., 2013), that are very high in some examples (Ma et al., 2009; Muenhor et al., 2010), the levels are low in comparison to indoor dusts in the UK, homes and offices and cars which residents are exposed to every day (Harrad et al., 2008). • Carcinogenic and non-carcinogenic risk not investigated through this pathway.	• Results may be confounded with open burning and smelting that may take place in the same location. • Limited occupational exposure data in LIMICs	• Adults and children have no choice to avoid exposure if they live around e-waste dismantling activities.	1	4	**4**	LIMIC
BFR	Ingestion of dust, soil and water	E-waste workers (informal)	NGA, CHN, THA	• Non-carcinogenic risk determined by one study (Ohajinwa et al., 2019b) and considered low in all but one case involving dust. • Carcinogenic risk low in all cases reported in one study (Ohajinwa et al., 2019b).		• Low hygiene means informal workers at risk of ingesting, particularly dust that they come into contact with.	1	4	**4**	LIMIC
		Population		• Though concentrations have been determined in soils and dusts in surrounding e-waste dismantling areas (Chen et al., 2018; Ohajinwa et al., 2019b; Wang et al., 2013), that are very high in some examples (Ma et al., 2009; Muenhor et al., 2010), the levels are low in comparison to indoor dusts in the UK, homes and offices and cars which residents are exposed to every day (Harrad et al., 2008). • Carcinogenic and non-carcinogenic risk not investigated through this pathway.	• Results may be confounded with open burning and smelting that may take place in the same location. • Limited occupational exposure data in LIMICs	• Adults and children have no choice to avoid exposure if they live around e-waste dismantling activities.	1	4	**4**	LIMIC
BFR	Dermal contact with soil and dust	E-waste workers (informal)	NGA, CHN, THA	• Though concentrations have been determined in soils and dusts in surrounding e-waste dismantling areas (Chen et al., 2018; Ohajinwa et al., 2019b; Wang et al., 2013), that are very high in some examples (Ma et al., 2009; Muenhor et al., 2010),		• Informal workers operate without dermal protective equipment.	3	4	**12**	LIMIC

Haz.	Pathway	Receptor	Geog.	Evidence and justification for risk assessment	Uncertainty (aleatoric and epistemic)	Receptor vulnerability	L	S	R	Global receptor context
				the levels are low in comparison to indoor dusts in the UK, homes and offices and cars which residents are exposed to every day (Harrad et al., 2008). • Non-carcinogenic risk calculated to be high in most cases (Ohajinwa et al., 2019b). • Carcinogenic risk considered high in some cases of exposure to dust (Ohajinwa et al., 2019b).						

Abbreviations: Abbreviations: likelihood (L); severity (S); risk (R); hazard being assessed (Haz.); low income and middle income countries (LIMIC); high income countries (HICs); geographical context (Geog.); potentially toxic elements (PTE); Brominated flame retardants (BFR); personal protective equipment (PPE); low income and middle income countries (LIMICs); cathode ray tube (CRT).

4. Challenge 2: Hydrometallurgical treatment to recover metals

4.1. Context

As well as the major constituent metals, such as steel and copper, e-waste contains several precious metals, including: platinum (Pt), selenium (Se), gallium (Ga), gold (Au), palladium (Pd), tellurium (Te), silver (Ag), germanium (Ge), and tantalum (Ta) (Abdelbasir et al., 2018). These metals exist in electronic products and components as substances that are coated onto, or bonded to other substances and materials whose combined purpose is to conduct, resist or store electricity.

The bonded state of these metals within e-waste presents a challenge for those who wish to recover them. They cannot be simply prised out or physically abraded, meaning that thermal (pyrometallurgical) or liquid (hydrometallurgical) techniques must be used for separation. Pyrometallurgical treatment (heating) has been discussed by Cook et al. (2020b) and therefore, this section will focus on hydrometallurgical treatment which involves the treatment of e-waste with acids and alkalis to leach and recover metals. A further process, biometallurgical treatment, is also emerging as a potential technique, however at the time of writing this is thought to be at laboratory level of technological readiness (Abdelbasir et al., 2018) and is therefore not discussed here.

4.2. Types of hydrometallurgical process

Compared to pyrometallurgical treatment, hydrometallurgy is reported to be a more ‘exact’ method of e-waste metal recovery, as it is more readily controllable and doesn’t require emission control (Hsu et al., 2019). However, this assertion implies that the chemicals used in hydrometallurgical treatment are disposed of in a way which does not involve their uncontrolled release into soil or water-bodies.

There are many different types of hydrometallurgical treatment, many of which have been reviewed comprehensively for formal sector operations by Khaliq et al. (2014), Cui and Zhang (2008) and Abdelbasir et al. (2018). Simply, hydrometallurgy takes place in two steps:

(1) Comminuted circuitry is immersed in a leaching solution (lixiviant) that may include: nitric acid, hydrochloric acid, aqua regia (nitric and hydrochloric acids mixed at a molar ratio of about 1:3), sulphuric acid, cyanide, or alkalis such as ammonia. Metal are dissolved into the solution which can be strained from the solids where the non-metals and non-target metals remain (Abdelbasir et al., 2018).

(2) Metals are recovered from the dissolved solution by precipitation, electrorefining, electrowinning, ion exchange, solvent extraction, adsorption and cementation. The newly recovered solids then need to be dried out and are often melted into ingots for onward sale (Abdelbasir et al., 2018).

Looking at the list of lixiviants described in 1, it should be obvious that extreme care is required to prevent these liquids from coming into contact with humans and also to control the risk of discharge into the environment; each has the potential for toxicity and corrosion. In HICs, the use of these substances is generally tightly controlled by robust and strictly enforced legislation, however the following sections reveal evidence of activity in LIMICs where the health and wellbeing of people and protection of the environment are lacking in some hydrometallurgical treatment processes. If managed poorly or not managed at all, there a several pathways through which receptors may be exposed to three substances and residues from hydrometallurgical treatment which are summarise in the conceptual model in **Figure 4**.

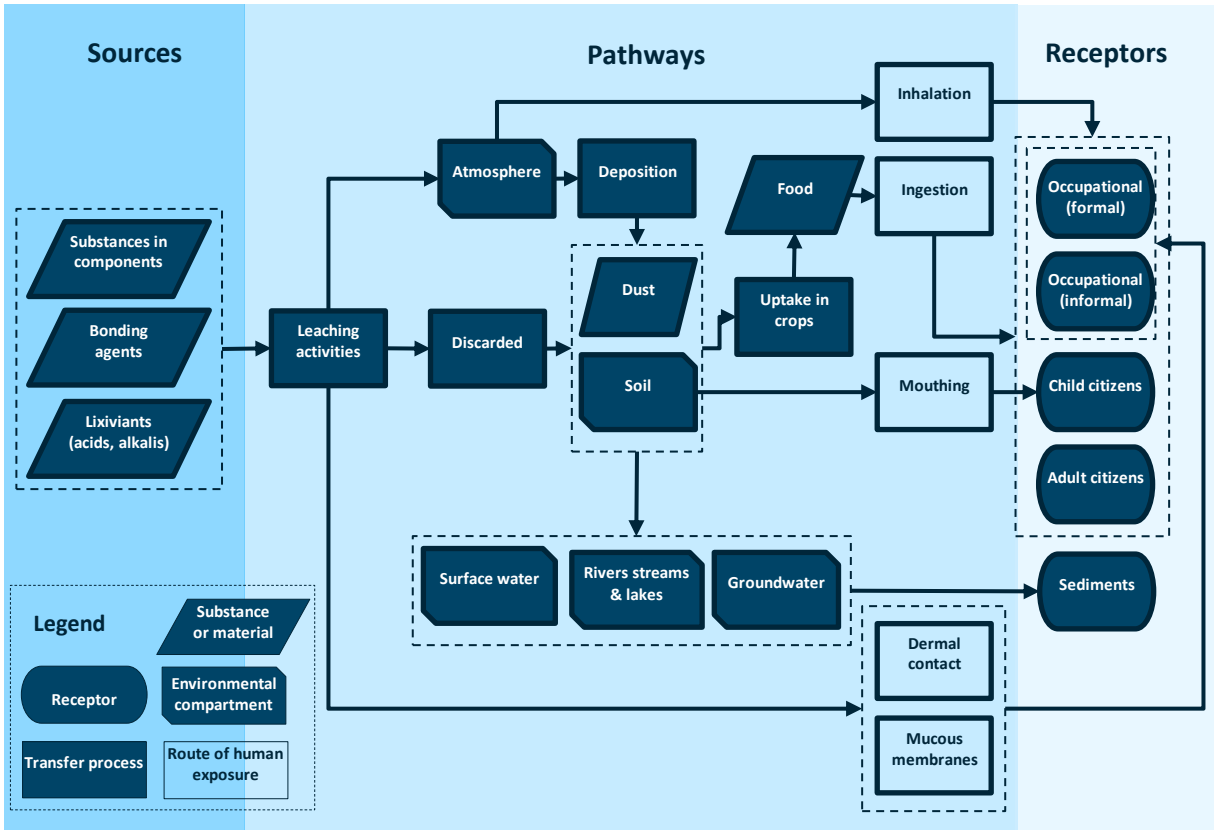


Figure 4: Hazard exposure conceptual model (source–pathway–receptor) associated with the hydrometallurgical treatment of e-waste.

There are tens, possibly hundreds of potential hydrometallurgical processes available to remove metals from e-waste (Abdelbasir et al., 2018; Cui and Zhang, 2008), though this

review finds little information on the specific types of processes used by the informal waste reclamation sector. In many publications reviewed here, references to hydrometallurgy are non-specific, mentioning acids, alkalis and other solvents. There are very few which directly observe and report on the particular reagents used, which presents a challenge for assessing the risk to occupational and public health and safety. Two lixiviants, cyanide and aqua regia have been reported to be prevalent (Puckett et al., 2002; Song and Li, 2014) and will therefore be expanded upon in **Sections 4.2.2** and **4.2.1** respectively.

4.2.1. Cyanide

Cyanide leaching has been used for the recovery of precious metals, like gold (Au), for more than 100 years (Syed, 2012) and is also effective at leaching metals, such as silver (Ag) and copper (Cu). Cyanide is relatively cheap to use and very effective; alternatives have been developed but they are less efficient and more expensive.

Keller (2006) described the process whereby potassium or sodium cyanide is dissolved in hot water and added to the gold containing circuitry which dissolves the gold at pH of around 10.5 to 12. Cementation of the pregnant solution is carried out by addition of a silver salt and aluminium foil, precipitating the gold and some of the silver. The solids are then heated and finally the gold and silver separated by nitric acid, which causes the gold to precipitate at approximately 200 g t^{-1} of printed circuit boards; Petter et al. (2014) reported recovery rates of 500 g t^{-1} from mobile phone circuitry – but the context was not provided, so it may be a formal or informal operation.

Twenty facilities were reported by Keller (2006) to be operating in the Bangalore area in India using cyanide leaching, one of which was investigated and found to have three workers of between 10 and 20 years of age. Japan International Cooperation Agency (2014) reported that cyanide leaching was used in the Philippines, but although cyanide is mentioned by most authors who discuss hydrometallurgical recovery from e-waste, there is no evidence to indicate the prevalence of this activity.

Cyanide compounds disrupt respiration of cells in the human body and are thus potentially toxic and life threatening (Jaszczak et al., 2017). Inhalation of cyanide gas (hydrogen cyanide) is the most dangerous form of exposure, however ingestion and dermal contact can also result in very serious poisoning and long term irreversible health implications (Centers for Disease Control and Prevention, 2018). Working with cyanide outdoors can lessen the

chance of inhalation, as can the use of respiratory protective equipment. However, as reported by Keller (2006), these control measures were absent in the facilities investigated in Bangalore. Several non-cyanide leaching reagents have been proposed in recent years, such as thiourea and thiosulfates, though they have yet to be adopted by industry (Abdelbasir et al., 2018).

4.2.2. Aqua regia (halide leaching)

Aqua regia is a mixture of nitric and hydrochloric acids at a molar ratio of about 1:3. Puckett et al. (2002) observed its use among informal e-waste reclaimers as a lixiviant used to recover gold from printed circuit boards in Guangzhou, an area of China notorious for informal e-waste recovery. Once the printed circuit boards have been stripped of components and physically removable solder, they are crushed and bathed in the solution which is then treated with a precipitant – possibly ferrous sulphate (Sheng and Etsell, 2007).

Puckett et al. (2002) observed clouds of steamy gases, most likely containing a combination of vaporised acids, nitrogen oxide and chlorine vapours (Schluep et al., 2009). Additionally, they reported river banks that had been turned black from the process sludge. Soil samples tested nearby showed a pH of 0.

Aqua regia use by the informal reclamation sector is also reported in China by Japan International Cooperation Agency (2014), however there was no further detail, beyond its stated use. Other studies detailed in **Section 4.4** indicate 'acid leaching and 'washing', but details are not provided so it is difficult to determine the reagents and activities that give rise to the observed concentrations of various substances in environmental compartments. It should also be noted that many other halide leaching processes exist, however aqua regia is focused on here as it is the primary method reported to be used by the informal recycling sector.

4.3. Occupational exposure

No reliable data was found in this review to indicate hazard exposure to chemical reagents used in hydrometallurgy. Puckett et al. (2002) reported that the informal hydrometallurgists observed wore gloves and wellington boots, but no other personal protective equipment and Keller (2006) observed no PPE use at all among workers, including children working with cyanide in Bangalore, India. Based on these few observations, it is hard to imagine that injuries do not take place, including chemical burns, poisoning through ingestion and

inhalation, and loss of sight, however there is no empirical evidence to support these speculations. Given the potentially hazardous nature of the substances used in hydrometallurgy it is a recommendation of this research that further study is carried out to determine the level of harm being caused to those participating in the activity.

4.4. Environmental media

Several studies reviewed here indicate discharge of untreated hydrometallurgical residues to the environment, often directly into water-bodies. As well as their direct toxicity to plant and animal life, both acids and cyanides carry with them residues of dissolved or suspended PTEs which can enter the water, be deposited into sediments and ingested by animals (Sepúlveda et al., 2010). Moreover, the clouds of acidic vapour observed by Puckett et al. (2002) may lead to deposition of condensed acidic vapour into the surrounding environment.

There are relatively few studies that specifically determine concentrations of potentially hazardous substances in environmental compartments as a consequence of hydrometallurgical treatment. Part of the reason for this may be that pyrometallurgical (thermal) treatment often takes place in the same locations and researchers may not want to communicate or may be prevented from communicating with informal workers, and therefore cannot always determine the activity that is taking place.

In **Table 21,** three studies are summarised that reported concentrations of PTEs in the environment surrounding acid washing operations in China (Chakraborty et al., 2019; Leung et al., 2007) and India (Chakraborty et al., 2019). Specifically, Wong et al. (2007) determined the concentration and distribution patterns of dissolved PTEs in aquatic systems close to an e-recycling site in China and tried to identify their sources related to primitive e-recycling activities. Excessive amounts of specific dissolved PTEs, including Ag, Cd, Co, Cu, Mo, Ni and Zn, were reported in river samples collected adjacent to an intensive acid leaching operation compared to other river samples close to the e-recycling site (Quan et al., 2015). This finding evidenced that acid washing of e-waste is a considerable source of dissolved PTEs leading to their accumulation in water bodies along related e-waste processes (Wong et al., 2007). The use of these aquatic systems and nearby areas for agricultural purposes posed a high concern to the local community as well (Wong et al., 2007). Similar conclusions were drawn by Quan et al. (2015), reporting elevated concentrations of Ni, Cu, Zn, Cd, Pb, Sn and Sb in the soil of informal acid leaching areas.

Table 21: Concentration of potentially toxic elements (PTEs) detected in environmental compartments in e-waste acid washing processes in informal sector.

Ref.	Geog.	Activity content	Media	n	Units	Ag	Cd	Co	Cu	Mo	Ni	Pb	Sb	Sn	Zn
		E-recycling site		8		0.059-0.303	0.260-0.362	2.79-4.94	39.7-67.3	1.39-2.27	43.3-66	1.33-2.24	16.3-22.1		89.9-117
Wong et al. (2007)	CHN	Acid washing	River	2	$\mu g\ l^{-1}$	0.335-0.354	0.540-0.554	4.98-5.08	85.5-89.7	2.32-2.35	92-94	1.73-1.87	21.1-21.4		122-128
		Acid washing		10			1.3	13.9	777		285	125	997	1.72	416
		Nearby paddy field		8			0.19	9.4	88		28.6	65	16.9	12.5	61.8
		Nearby deserted field		5			1.9	12.1	138		48.1	128	49.4	36.8	261
Quan et al. (2015)	CHN	Reference area	Soil	5	$mg\ kg^{-1}$		0.4	3.78	23		12.8	61.3	2.61	8.91	116
		Acid washing		5			0.3	11	1,931		15	193			
		Dismantling		5			3	7	273		75	197			
		Shredding		4			1	3	533		234	87			
Chakraborty et al. (2019)	IND	Open MSW burning	Soil	11	$mg\ kg^{-1}$		1	2	602		6	30			

Abbreviations: geographical context (Geog.); number of samples (n); municipal solid waste (MSW).

A recent study conducted in an informal recycling site in India reported excessive amounts only for Cu and Co compared to other e-recycling processes, although the sampling points for acid washing areas included also plastic and wire burning activities (Chakraborty et al., 2019), the likely reason that elevated levels of PAHs were also found. In addition to Chakraborty et al. (2019), two further studies were identified that investigated pollution levels in e-waste recovery workshops in China (**Table 22**). Leung et al. (2007) found elevated levels of PBDEs in the 'acid leaching area' and Leung et al. (2015) found elevated levels of PAHs in the same location several years later. However, Leung et al. (2007) also found high concentrations of dioxins and related compounds (DRCs) in these same areas, which can only have originated from the combustion of materials containing halogenated com pounds, for instance polyvinyl chloride (PVC) or plastics containing brominated flame retardants. The presence of these DRCs calls into question the source of both the PAHs and PBDEs observed by Leung et al. (2015) and Leung et al. (2007) respectively as they may just have easily occurred as a result of combustion as hydrometallurgical treatment. This exemplifies a potential shortcoming of many of the publications on this issue in that the activity stated for a particular area under investigation may not always be accurately reported, creating a challenge to determine which processes result in the greatest environmental or human exposure.

Table 22: Concentration of pollutants detected in environmental compartments in e-waste acid washing processes including plastics and wire burning in informal sector.

							Total dioxins			
Ref.	**Geog.**	**Activity content**	**Media**	**n**	**Units**	**PBDE**	**PCDD/Fs**	**I-TEQ**	**PAH**	**PCBs**
Leung et al. (2007)	CHN	Acid leaching area	Soil	3	ng g^{-1}	3,570	39.3	0.51		
		E-dumping area		3		1,440	0.55	0.0049		
		Nearby duck pond		3		398	7.36	0.034		
		Nearby rice field		3		48.2	2.73	0.0117		
Leung et al. (2015)	CHN	Acid leaching area	Soil	7	ng g^{-1}				1,950-5,210	
		E-dumping area		5					268	
		Nearby rice field area		9					457-171	
		Reference area		5					89-152	
Chakraborty et al. (2019)	IND	Acid washing	Soil	5	ng g^{-1}				2,198	148
		Dismantling		5					866	6.5
		Shredding		4					576	8.2
		Open MSW burning		11					1,029	3.4

Abbrev.: polybrominated diphenyl ethers (PBDE); polychlorinated dibenzo-p-dioxins and polychlorinated dibenzo-p-furans (PCDD/Fs; international toxic equivalency factor (I-TEQ); polycyclic aromatic hydrocarbons (PAHs); polychlorinated biphenyls (PCB); geographical research context (Geog.); number of samples (n); municipal solid waste (MSW).

4.1. Risk characterisation for hydrometallurgical treatment of e-waste

The limited observational evidence for occupational and public risk resulting from the hydrometallurgical treatment of e-waste presented a considerable challenge when carrying out the semi-quantitative risk assessment shown in **Table 23**. Only two studies observed activities involving cyanide and aqua regia, despite the practices being widely reported to exist. Though the magnitude of this type of small-scale informal hydrometallurgical treatment isn't well understood, the few observed circumstances alongside observations of safety levels observed elsewhere in the informal sector (Cook and Velis, 2020b), infer plausibility for poor practices throughout the e-waste reprocessing sector across LIMICs. The close handling of large quantities of cyanide by children is extremely concerning given the potential life threatening and irreversible morbidity risk. Equally concerning is the handling of low pH substances without eye or face protection alongside its subsequent discharge into the environment surrounding small scale hydrometallurgical activities. Risks posed by PTEs as a result of hydrometallurgy are less certain but still worthy of concern and have been scored as posing a low to medium risk in our semi-quantitative assessment. Although the likelihood of dissolved and suspended PTEs in hydrometallurgical agents is reported, there are also many confounding activities that may have been the source in the studies that reported them.

Table 23: Risk characterisation summary for hydrometallurgical treatment of e-waste (non-thermal).

Haz.	Pathway	Receptor	Geog.	Evidence and justification for risk assessment	Uncertainty (aleatoric and epistemic)	Receptor vulnerability	L	S	R	Global receptor context
Cyanide	Atmosphere/ inhalation, dermal contact, ingestion	E-waste workers (informal)	IND	• Hydrometallurgical treatment using cyanide observed to be carried out informally by children and adults in India without PPE (Keller, 2006). • Potential for very serious poisoning and long term irreversible health implications (Centers for Disease Control and Prevention, 2018; Jaszczak et al., 2017).	• Only one observational study with inferred activity reported by other authors without evidence – inherently problematic to observe these practices.	• Children have less risk awareness and understanding of the consequences. • Physiology of children more vulnerable to poisoning. • Though processing carried out in open air, complete lack of PPE makes receptors more vulnerable to exposure.	4	5	20	LIMICs
		E-waste workers (Children)								
		Population					4	4	16	
Aqua regia	Atmosphere/ inhalation, dermal contact (burns), ingestion	E-waste workers (informal)	CHN	• Hydrometallurgical treatment using aqua regia observed to be carried out by adults in China (Puckett et al., 2002), and reported (not observed) by Japan International Cooperation Agency (2014). • Puckett et al. (2002) observed use of aqua regia with gloves and wellingtons, but eye and face protection was not mentioned. • Evidence of discharge directly into surrounding environment (Chakraborty et al., 2019; Leung et al., 2007; Wong et al., 2007).	• Only one observational study with inferred activity reported by other authors without evidence – inherently problematic to observe these practices.	• Given the limited PPE used (only hands and feet), it is hard to imagine that injuries do not take place.	4	5	20	LIMICs
		Population					4	4	16	
PTE	Dermal contact / ingestion	E-waste workers (informal)	CHN, IND	• Pathway for dissolved and suspended PTEs from hydrometallurgical processes established (Sepúlveda et al., 2010). • Acid washing of e-waste is evidenced to be a source of dissolved PTEs leading to their accumulation in water bodies along related e-waste processes (Chakraborty et al., 2019; Leung et al., 2007; Wong et al., 2007).	• Results may be confounded with open burning, smelting and physical processing that may take place in the same location.	• Informal workers operate without dermal protective equipment.	2	4	8	LIMICs
		Population				• Adults and children have no choice to avoid exposure if they live around e-waste dismantling activities.	2	4	8	
BFR, PAH, PCB	Dermal contact / ingestion	Population / workers (informal)	CHN, IND	• Correlation between acid washing areas of e-waste reprocessing sites and high concentrations of BFRs, PCBs and PAHs (Chakraborty et al., 2019; Leung et al., 2015; Leung et al., 2007).	• High probability that results confounded with open burning activities that take place in the same area as acid washing indicated by concentrations of DRCs found in the same area.	• Adults and children have no choice to avoid exposure if they live around e-waste dismantling activities.	na	na	na	LIMICs

Abbreviations: likelihood (L); severity (S); risk (R); hazard being assessed (Haz.); low income and middle income countries (LIMIC); geographical context (Geog.); potentially toxic elements (PTE); Brominated flame retardants (BFR); polycyclic aromatic hydrocarbons (PAH); polychlorinated biphenyls (PCB); personal protective equipment (PPE).

Though there is some evidence of a correlation between BFRs, PAHs and PCBs in environmental medial close to acid washing sites, there were clearly confounding factors evidenced by the presence of DRCs that could only have originated from combustion, resulting in these hazard-pathway-receptor combinations being unscored.

5. Conclusion

Despite efforts to curb the transboundary movement of e-waste, exports from HICs to LIMICs continue, providing feedstock to informal and unregulated reclamation yards throughout the Global South. Here, we have systematically identified and arranged data from 36 core sources, demonstrating that a large range of hazardous and potentially toxic substances are emitted when e-waste is physically and hydrometallurgically processed, sometimes by children or in close proximity to their settlements. Many of the workers who carry out e-waste reclamation activities do so without protective equipment or any formal safe system of work, leaving them vulnerable to exposure, morbidity and potentially early mortality. In addition to substances that may be emitted from the e-waste components and assemblies during processing, some highly hazardous substances are used in the hydrometallurgical processes to dissolve, separate and recover valuable metals that are bonded into electronic composite matrices. Alarming evidence exists for extensive use of cyanide by children in India, and variations of aqua regia, a low pH oxidising mixture, believed to be used throughout the informal e-waste reclamation sector worldwide.

The level of information available on informal hydrometallurgical treatment is limited. Given the potential hazardousness and toxicity of the substances being used, it is improbable that accidents and serious injuries do not occur. However, the data paucity on this subject presents a considerable challenge in determining the prevalence and actual health effects experiences from hydrometallurgical treatment, especially as these informal options are inherently under-reported.

Though a large body of research already exists to evidence the hazardous emissions from e-waste, there are still challenges to be overcome in linking observed concentrations in the environment to specific activities and thus determining causality. For instance, we have revealed several examples where concentrations of substances in environmental media may have been erroneously linked to acid washing activities, when there is a strong inference that combustion practices may also be involved. The co-location of combustion, hydrometallurgy

and physical processing in many informal e-waste reclamation sites will continue to present this challenge. Determination of the most damaging activities is critical if interventions are to be designed to transition to safer practices. Perhaps one of the most perplexing factors is that informal e-waste recycling is considered illicit, and thus may continue to operate without oversight from regulators, furthering the challenge. As discussed by Velis et al. (2012), inclusion and integration of the informal recycling sector into national and municipal waste management plans could provide a platform from which to develop links and improve the lives of those who are effected by the hazardous and dangerous aspects of e-waste physical and hydrometallurgical processing.

CRediT author statement

Ed Cook: Conceptualization; Data curation; Formal analysis; Investigation; Methodology; Project administration; Resources; Validation; Visualization; Writing – original draft; Writing – review & editing. **Costas A. Velis:** Conceptualization; Data curation; Formal Analysis; Funding acquisition; Investigation; Methodology; Project administration; Resources; Software; Supervision; Validation; Visualization; Writing – original draft; Writing – review & editing. **Spyridoula Gerasimidou**: Data curation; Investigation; Writing – original draft; Writing – review & editing; **Aditi Ramola**: Formal analysis; Funding acquisition; Project administration; Resources; Supervision; Visualization; Writing – original draft. **Arne Ragossnig**: Writing – original draft; Funding acquisition; Supervision; Project administration.

Acknowledgements

We are grateful to the Technical Advisory Board of the Engineering X Safer End of Engineered Life programme, of the Royal Academy of Engineering for their steering and insightful feedback, especially on early versions of this research and manuscript. We thank the Programme Board, chaired by Professor William Powrie FREng & the Academy staff, especially Hazel Ingham and Shaarad Sharma who provided support throughout the process. Ad hoc advice, guidance and criticism was provided by multiple stakeholder representatives, as listed in the relevant Engineering X report. We are grateful to Nick Rigas, (D-Waste) for the presentation of infographics and to Boris Brkovic (ISWA) for support with preliminary data collection. The research communicated and opinions expressed here are authors' alone.

Financial

This work was made possible by the Engineering X Safer End of Engineered Life programme which is funded by Lloyd's Register Foundation. Engineering X is an international collaboration, founded by the Royal Academy of Engineering and Lloyd's Register Foundation, that brings together some of the world's leading problem-solvers to address the great challenges of our age. The Engineering X Safer End of Engineered Life programme which seeks to improve safety and reduce harm caused by the decommissioning, dismantling and disposal of engineered products, artefacts, and structures at the end of their life.

References

Abdelbasir, S.M., Hassan, S.S., Kamel, A.H. and El-Nasr, R.S. (2018). Status of electronic waste recycling techniques: a review. *Environmental Science and Pollution Research*, 25(17), 16533-16547.

Ackah, M. (2017). Informal E-waste recycling in developing countries: review of metal(loid)s pollution, environmental impacts and transport pathways. *Environmental Science and Pollution Research*, 24(31), 24092-24101.

Ackah, M. (2019). Soil elemental concentrations, geoaccumulation index, non-carcinogenic and carcinogenic risks in functional areas of an informal e-waste recycling area in Accra, Ghana. *Chemosphere*, 235, 908-917.

Amankwaa, E.F., Adovor Tsikudo, K.A. and Bowman, J. (2017). 'Away' is a place: The impact of electronic waste recycling on blood lead levels in Ghana. *Science of The Total Environment*, 601-602, 1566-1574.

Asadi Dalini, E., Karimi, G., Zandevakili, S. and Goodarzi, M. (2020). A Review on environmental, economic and hydrometallurgical processes of recycling spent lithium-ion batteries. *Mineral Processing and Extractive Metallurgy Review*, 10.1080/08827508.2020.1781628, 1-22.

Asante, K.A., Agusa, T., Biney, C.A., Agyekum, W.A., Bello, M., Otsuka, M., Itai, T., Takahashi, S. and Tanabe, S. (2012). Multi-trace element levels and arsenic speciation in urine of e-waste recycling workers from Agbogbloshie, Accra in Ghana. *Science of The Total Environment*, 424, 63-73.

Awasthi, A.K. and Li, J. (2017). Management of electrical and electronic waste: A comparative evaluation of China and India. *Renewable and Sustainable Energy Reviews*, 76, 434-447.

Awasthi, A.K., Wang, M., Wang, Z., Awasthi, M.K. and Li, J. (2018). E-waste management in India: A mini-review. *Waste Management & Research*, 36(5), 408-414.

Awasthi, A.K., Zeng, X. and Li, J. (2016). Environmental pollution of electronic waste recycling in India: A critical review. *Environmental Pollution*, 211, 259-270.

Awere, E., Obeng, P.A., Bonoli, A. and Obeng, P.A. (2020). E-waste recycling and public exposure to organic compounds in developing countries: a review of recycling practices and toxicity levels in Ghana. *Environmental Technology Reviews*, 9(1), 1-19.

Bimir, M.N. (2020). Revisiting e-waste management practices in selected African countries. *Journal of the Air & Waste Management Association*, 70(7), 659-669.

Borthakur, A. and Govind, M. (2018). Public understandings of e-waste and its disposal in urban India: From a review towards a conceptual framework. *Journal of Cleaner Production*, 172, 1053-1066.

Brigden, K., Labunska, I., Santillo, D. and Allsopp, M. (2005). *Recycling of electronic wastes in China & India: Workplace & environmental contamination*. Greenpeace International. Retrieved from: https://storage.googleapis.com/planet4-international-stateless/2005/08/ee56bf32-recycling-of-electronic-waste.pdf.

Brigden, K., Labunska, I., Santillo, D. and Johnston, P. (2008). *Chemical contamination at e-waste recycling and disposal sites in Accra and Korforidua, Ghana*. Amsterdam, The Netherlands: Greenpeace International. Retrieved from: https://www.greenpeace.to/publications/chemical-contamination-at-e-wa.pdf.

Bruce-Vanderpuije, P., Megson, D., Reiner, E.J., Bradley, L., Adu-Kumi, S. and Gardella, J.A. (2019). The state of POPs in Ghana- A review on persistent organic pollutants: Environmental and human exposure. *Environmental Pollution*, 245, 331-342.

Burns, C.J., LaKind, J.S., Mattison, D.R., Alcala, C.S., Branch, F., Castillo, J., Clark, A., Clougherty, J.E., Darney, S.P., Erickson, H., Goodman, M., Greiner, M., Jurek, A.M., Miller, A., Rooney, A.A. and Zidek, A. (2019). A matrix for bridging the epidemiology and risk assessment gap. *Global Epidemiology*, 1, 100005.

Cahill, T.M., Groskova, D., Charles, M.J., Sanborn, J.R., Denison, M.S. and Baker, L. (2007). Atmospheric concentrations of polybrominated diphenyl ethers at near-source sites. *Environmental Science & Technology*, 41(18), 6370-6377.

Cai, H., Xu, X., Zhang, Y., Cong, X., Lu, X. and Huo, X. (2019). Elevated lead levels from e-waste exposure are linked to sensory integration difficulties in preschool children. *NeuroToxicology*, 71, 150-158.

Cao, P., Fujimori, T., Juhasz, A., Takaoka, M. and Oshita, K. (2020). Bioaccessibility and human health risk assessment of metal (loid) s in soil from an e-waste open burning site in Agbogbloshie, Accra, Ghana. *Chemosphere*, 240, 124909.

Ceballos, D., Beaucham, C. and Page, E. (2017). Metal Exposures at three US electronic scrap recycling facilities. *Journal of Occupational and Environmental Hygiene*, 14(6), 401-408.

Centers for Disease Control and Prevention (2018) *Emergency Preparedness and Response: Facts About Cyanide.* Retreived from https://emergency.cdc.gov/agent/cyanide/basics/facts.asp#:~:text=Cyanide%20prevents%20the%20cells%20of,use%20a%20lot%20of%20oxygen.

Centers for Disease Control and Prevention (2019) *Blood lead levels in children.* Retreived from https://www.cdc.gov/nceh/lead/prevention/blood-lead-levels.htm.

Chakraborty, P., Sampath, S., Mukhopadhyay, M., Selvaraj, S., Bharat, G.K. and Nizzetto, L. (2019). Baseline investigation on plasticizers, bisphenol A, polycyclic aromatic hydrocarbons and heavy metals in the surface soil of the informal electronic waste recycling workshops and nearby open dumpsites in Indian metropolitan cities. *Environmental Pollution*, 248, 1036-1045.

Chen, A., Dietrich, K.N., Huo, X. and Ho, S.m. (2010). Developmental neurotoxicants in e-waste: an emerging health concern. *Environmental Health Perspectives*, 119(4), 431-438.

Chen, H., Lam, J.C.W., Zhu, M., Wang, F., Zhou, W., Du, B., Zeng, L. and Zeng, E.Y. (2018). Combined effects of dust and dietary exposure of occupational workers and local residents to short- and medium-chain chlorinated paraffins in a mega e-waste recycling industrial park in south China. *Environmental Science and Technology*, 52(20), 11510-11519.

Cook, E. and Velis, C.A. (2020a). Construction and demolition waste management: A systematic review of risks to occupational and public health. *engrXiv*, 10.31224/osf.io/5tpbz.

Cook, E. and Velis, C.A. (2020b). *Global review on safer end of engineered life*. London, UK: Royal Academy of Engineering. Retrieved from: https://doi.org/10.5518/100/58.

Cook, E., Velis, C.A. and Derks, M. (2020a). Plastic waste reprocessing for circular economy: A systematic review of risks to occupational and public health from legacy substances and extrusion. *engrXiv*, 10.31224/osf.io/yxb5u.

Cook, E., Velis, C.A., Gerassimidou, S., Ramola, A. and Ragossnig, A. (2020b). Thermal deconstruction, open burning and disposal of e-waste without pollution control: A systematic review of risks to occupational and public health. *engrXiv*, 10.31224/osf.io/tbrmq.

Cook, E., Velis, C.A., Woolridge, A., Stapp, P. and Edmondson, S. (2020c). Medical and healthcare waste generation, storage, treatment and disposal: A systematic review of risks to occupational and public health. *engrXiv*, 10.31224/osf.io/tb7ng.

Cooper, C., Booth, A., Varley-Campbell, J., Britten, N. and Garside, R. (2018). Defining the process to literature searching in systematic reviews: a literature review of guidance and supporting studies. *BMC Medical Research Methodology*, 18(1), 85.

Cui, J. and Zhang, L. (2008). Metallurgical recovery of metals from electronic waste: A review. *Journal of Hazardous Materials*, 158(2), 228-256.

Damrongsiri, S., Vassanadumrongdee, S. and Tanwattana, P. (2016). Heavy metal contamination characteristic of soil in WEEE (waste electrical and electronic equipment) dismantling community: a case study of Bangkok, Thailand. *Environmental Science and Pollution Research*, 23(17), 17026-17034.

Daum, K., Stoler, J. and Grant, R.J. (2017). Toward a more sustainable trajectory for e-waste policy: A review of a decade of e-waste research in Accra, Ghana. *International Journal of Environmental Research and Public Health*, 14(2).

Duan, H., Miller, T.R., Gregory, J., Kirchain, R. and Linnell, J. (2013). *Quantitative characterization of domestic and transboundary flows of used electronics: Analysis of generation, collection, and export in the United States*. StEP. Retrieved from: http://www.step-initiative.org/files/_documents/other_publications/MIT-NCER%20US%20Used%20Electronics%20Flows%20Report%20-%20December%202013.pdf.

Efsa Panel on Contaminants in the Food Chain, Schrenk, D., Bignami, M., Bodin, L., Chipman, J.K., del Mazo, J., Grasl-Kraupp, B., Hogstrand, C., Hoogenboom, L., Leblanc, J.C., Nebbia, C.S., Ntzani, E., Petersen, A., Sand, S., Schwerdtle, T., Vleminckx, C., Wallace, H., Brüschweiler, B., Leonards, P., Rose, M., Binaglia, M., Horváth, Z., Ramos Bordajandi, L. and Nielsen, E. (2020). Risk assessment of chlorinated paraffins in feed and food. *EFSA Journal*, 18(3), e05991.

Egorova, K.S. and Ananikov, V.P. (2017). Toxicity of Metal Compounds: Knowledge and Myths. *Organometallics*, 36(21), 4071-4090.

European Union (2006). Commission Regulation (EC) No 1881/2006 of 19 December 2006 setting maximum levels for certain contaminants in foodstuffs In Off. J. Eur. Union (ed.) *L364, p5*.

European Union (2015). *Commission Regulation (EU) 2015/1006 of 25 June 2015 amending Regulation (EC) No 1881/2006 as regards maximum levels of inorganic arsenic in foodstuffs*. Off. J. Eur. Union. Retrieved from: https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:02006R1881-20100701&from=EN.

Fang, W., Yang, Y. and Xu, Z. (2013). PM10 and PM2.5 and health risk assessment for heavy metals in a typical factory for cathode ray tube television recycling. *Environmental Science and Technology*, 47(21), 12469-12476.

Fujimori, T., Takigami, H., Agusa, T., Eguchi, A., Bekki, K., Yoshida, A., Terazono, A. and Ballesteros Jr, F.C. (2012). Impact of metals in surface matrices from formal and informal

electronic-waste recycling around Metro Manila, the Philippines, and intra-Asian comparison. *Journal of Hazardous Materials*, 221, 139-146.

Glendon, A.I. and Stanton, N.A. (2000). Perspectives on safety culture. *Safety Science*, 34(1), 193-214.

Grant, K., Goldizen, F.C., Sly, P.D., Brune, M.-N., Neira, M., van den Berg, M. and Norman, R.E. (2013). Health consequences of exposure to e-waste: a systematic review. *The lancet global health*, 1(6), e350-e361.

Gravel, S., Aubin, S. and Labrèche, F. (2019a). Assessment of occupational exposure to organic flame retardants: A systematic review. *Annals of Work Exposures and Health*, 63(4), 386-406.

Gravel, S., Lavoue, J., Bakhiyi, B., Diamond, M.L., Jantunen, L.M., Lavoie, J., Roberge, B., Verner, M.A., Zayed, J. and Labreche, F. (2019b). Halogenated flame retardants and organophosphate esters in the air of electronic waste recycling facilities: Evidence of high concentrations and multiple exposures. *Environment International*, 128, 244-253.

Harrad, S., Ibarra, C., Abdallah, M.A.-E., Boon, R., Neels, H. and Covaci, A. (2008). Concentrations of brominated flame retardants in dust from United Kingdom cars, homes, and offices: Causes of variability and implications for human exposure. *Environment International*, 34(8), 1170-1175.

Herat, S. and Agamuthu, P. (2012). E-waste: a problem or an opportunity? Review of issues, challenges and solutions in Asian countries. *Waste Management & Research*, 30(11), 1113-1129.

Hong-Gang, N., Shao-You, L., Ting, M. and Hui, Z. (2016). Brominated flame retardant emissions from the open burning of five plastic wastes and implications for environmental exposure in China. *Environmental Pollution*, 214, 70-76.

Hopson, E. and Puckett, J. (2016). *Scam recycling: E-dumping on Asia by US recyclers*. Seattle, WA: Basel Action Network. Retrieved from: http://wiki.ban.org/images/1/12/ScamRecyclingReport-web.pdf.

Hsu, E., Barmak, K., West, A.C. and Park, A.H.A. (2019). Advancements in the treatment and processing of electronic waste with sustainability: a review of metal extraction and recovery technologies. *Green Chemistry*, 21(5), 919-936.

Huang, C.L., Bao, L.J., Luo, P., Wang, Z.Y., Li, S.M. and Zeng, E.Y. (2016). Potential health risk for residents around a typical e-waste recycling zone via inhalation of size-fractionated particle-bound heavy metals. *Journal of Hazardous Materials*, 317, 449-456.

Hunter, P.R., Payment, P., Ashbolt, N. and Bartram, J. (2003). Assessment of risk *Assessing microbial safety of drinking water: Improving approaches and methods* (79-109). Paris, France: Organisation for Economic Cooperation and Development and World Health Organisation.

Iozza, S., Müller, C.E., Schmid, P., Bogdal, C. and Oehme, M. (2008). Historical profiles of chlorinated paraffins and polychlorinated biphenyls in a dated sediment core from Lake Thun (Switzerland). *Environmental Science & Technology*, 42(4), 1045-1050.

Iqbal, M., Breivik, K., Syed, J.H., Malik, R.N., Li, J., Zhang, G. and Jones, K.C. (2015). Emerging issue of e-waste in Pakistan: A review of status, research needs and data gaps. *Environmental Pollution*, 207, 308-318.

Isimekhai, K.A., Garelick, H., Watt, J. and Purchase, D. (2017). Heavy metals distribution and risk assessment in soil from an informal E-waste recycling site in Lagos State, Nigeria. *Environmental Science and Pollution Research*, 24(20), 17206-17219.

Ismail, H. and Hanafiah, M.M. (2019). Discovering opportunities to meet the challenges of an effective waste electrical and electronic equipment recycling system in Malaysia. *Journal of Cleaner Production*, 238, 117927.

Iwegbue, C.M.A., Eyengho, S.B., Egobueze, F.E., Odali, E.W., Tesi, G.O., Nwajei, G.E. and Martincigh, B.S. (2019). Polybrominated diphenyl ethers and polychlorinated biphenyls in indoor dust from electronic repair workshops in southern Nigeria: Implications for onsite human exposure. *Science of The Total Environment*, 671, 914-927.

Japan International Cooperation Agency (2014). *Data collection survey on e-waste management in Malaysia and surrounding countries*. Japan International Cooperation Agency (JICA) EX Research Institute Ltd. Retrieved from: https://openjicareport.jica.go.jp/pdf/12154589.pdf.

Jaszczak, E., Polkowska, Ż., Narkowicz, S. and Namieśnik, J. (2017). Cyanides in the environment-analysis-problems and challenges. *Environmental science and pollution research international*, 24(19), 15929-15948.

Julander, A., Lundgren, L., Skare, L., Grander, M., Palm, B., Vahter, M. and Liden, C. (2014). Formal recycling of e-waste leads to increased exposure to toxic metals: An occupational exposure study from Sweden. *Environment International*, 73, 243-251.

Julander, A., Westberg, H., Engwall, M. and van Bavel, B. (2005). Distribution of brominated flame retardants in different dust fractions in air from an electronics recycling facility. *Science of The Total Environment*, 350(1), 151-160.

Kaya, G.K., Ward, J.R. and Clarkson, P.J. (2018). A framework to support risk assessment in hospitals. *International Journal for Quality in Health Care*, 31(5), 393-401.

Kaya, M. (2016). Recovery of metals and nonmetals from electronic waste by physical and chemical recycling processes. *Waste Management*, 57, 64-90.

Keller, M. (2006). *Assessment of gold recovery processes in Bangalore, India and evaluation of an alternative recycling path for printed wiring boards: a case study*. (Diploma ETH Zurich, Zurich, Germany) Retrieved from https://www.yumpu.com/en/document/read/33108068/assessment-of-gold-recovery-processes-in-bangalore-india-empa.

Khaliq, A., Rhamdhani, A.M., Brooks, G. and Masood, S. (2014). Metal extraction processes for electronic waste and existing industrial routes: a review and Australian perspective. *Resources*, 3(1).

Kim, Y.R., Harden, F.A., Toms, L.-M.L. and Norman, R.E. (2014). Health consequences of exposure to brominated flame retardants: A systematic review. *Chemosphere*, 106, 1-19.

Lee, D., Offenhuber, D., Duarte, F., Biderman, A. and Ratti, C. (2018). Monitour: Tracking global routes of electronic waste. *Waste Management*, 72, 362-370.

Lepawsky, J. and McNabb, C. (2010). Mapping international flows of electronic waste. *The Canadian Geographer / Le Géographe canadien*, 54(2), 177-195.

Leung, A.O., Cheung, K.C. and Wong, M.H. (2015). Spatial distribution of polycyclic aromatic hydrocarbons in soil, sediment, and combusted residue at an e-waste processing site in southeast China. *Environmental Science and Pollution Research*, 22(12), 8786-8801.

Leung, A.O., Luksemburg, W.J., Wong, A.S. and Wong, M.H. (2007). Spatial distribution of polybrominated diphenyl ethers and polychlorinated dibenzo-p-dioxins and dibenzofurans in soil and combusted residue at Guiyu, an electronic waste recycling site in southeast China. *Environmental Science & Technology*, 41(8), 2730-2737.

Liu, Y., Huo, X., Xu, L., Wei, X., Wu, W., Wu, X. and Xu, X. (2018). Hearing loss in children with e-waste lead and cadmium exposure. *Science of The Total Environment*, 624, 621-627.

Lu, C., Zhang, L., Zhong, Y., Ren, W., Tobias, M., Mu, Z., Ma, Z., Geng, Y. and Xue, B. (2015). An overview of e-waste management in China. *Journal of Material Cycles and Waste Management*, 17(1), 1-12.

Ma, J., Addink, R., Yun, S., Cheng, J., Wang, W. and Kannan, K. (2009). Polybrominated dibenzo-p-dioxins/dibenzofurans and polybrominated diphenyl ethers in soil, vegetation, workshop-floor dust, and electronic shredder residue from an electronic waste recycling facility and in soils from a chemical industrial complex in eastern China. *Environmental Science & Technology*, 43(19), 7350-7356.

Malkoske, T., Tang, Y., Xu, W., Yu, S. and Wang, H. (2016). A review of the environmental distribution, fate, and control of tetrabromobisphenol A released from sources. *Science of The Total Environment*, 569-570, 1608-1617.

McGrath, T.J., Ball, A.S. and Clarke, B.O. (2017). Critical review of soil contamination by polybrominated diphenyl ethers (PBDEs) and novel brominated flame retardants (NBFRs); concentrations, sources and congener profiles. *Environmental Pollution*, 230, 741-757.

Mmereki, D., Li, B. and Li'ao, W. (2015). Waste electrical and electronic equipment management in Botswana: Prospects and challenges. *Journal of the Air & Waste Management Association*, 65(1), 11-26.

Moher, D., Liberati, A., Tetzlaff, J., Altman, D.G. and The Prisma Group (2009). Preferred reporting items for systematic reviews and meta-analyses: The PRISMA statement. *PLOS Medicine*, 6(7), e1000097.

Morton, J., Tan, E., Leese, E. and Cocker, J. (2014). Determination of 61 elements in urine samples collected from a non-occupationally exposed UK adult population. *Toxicology Letters*, 231(2), 179-193.

Muenhor, D., Harrad, S., Ali, N. and Covaci, A. (2010). Brominated flame retardants (BFRs) in air and dust from electronic waste storage facilities in Thailand. *Environment International*, 36(7), 690-698.

National Research Council (US) Subcommittee on Flame-Retardant Chemicals (2000). Chlorinated paraffins *Toxicological risks of selected flame-retardant chemicals*. Washington (DC): National Academies Press (US).

NHMRC (2010). *Cancer risk assessment methodology: A review and recommendations*. Retrieved from: http://www.nepc.gov.au/system/files/pages/9b067155-4726-423b-989b-5263263b9c16/files/b4-cancer-methodology-final-draft-sep2010.pdf.

Nnorom, I.C. and Osibanjo, O. (2008a). Electronic waste (e-waste): Material flows and management practices in Nigeria. *Waste Management*, 28(8), 1472-1479.

Nnorom, I.C. and Osibanjo, O. (2008b). Overview of electronic waste (e-waste) management practices and legislations, and their poor applications in the developing countries. *Resources, Conservation and Recycling*, 52(6), 843-858.

Occupational Safety and Health Administration (2020) *Occupational Safety and Health Administration.* Retreived from https://www.osha.gov/.

Oguri, T., Suzuki, G., Matsukami, H., Uchida, N., Tue, N.M., Tuyen, L.H., Viet, P.H., Takahashi, S., Tanabe, S. and Takigami, H. (2018). Exposure assessment of heavy metals in an e-waste processing area in northern Vietnam. *Science of The Total Environment*, 621, 1115-1123.

Ohajinwa, C.M., Van Bodegom, P.M., Osibanjo, O., Xie, Q., Chen, J., Vijver, M.G. and Peijnenburg, W. (2019a). Health risks of polybrominated diphenyl ethers (PBDEs) and metals at informal electronic waste recycling sites. *International Journal of Environmental Research Public Health*, 16(6), 906.

Ohajinwa, C.M., van Bodegom, P.M., Osibanjo, O., Xie, Q., Chen, J.W., Vijver, M.G. and Peijnenburg, W. (2019b). Health risks of polybrominated diphenyl ethers (PBDEs) and metals at informal electronic waste recycling sites. *International Journal of Environmental Research and Public Health*, 16(6).

Ohajinwa, C.M., van Bodegom, P.M., Vijver, M.G. and Peijnenburg, W.J.G.M. (2018). Impact of informal electronic waste recycling on metal concentrations in soils and dusts. *Environmental Research*, 164, 385-394.

Ongondo, F.O., Williams, I.D. and Cherrett, T.J. (2011). How are WEEE doing? A global review of the management of electrical and electronic wastes. *Waste Management*, 31(4), 714-730.

Petter, P.M.H., Veit, H.M. and Bernardes, A.M. (2014). Evaluation of gold and silver leaching from printed circuit board of cellphones. *Waste Management*, 34(2), 475-482.

Pourret, O. and Hursthouse, A. (2019). It's time to replace the term "heavy metals" with "potentially toxic elements" when reporting environmental research. *International Journal of Environmental Research and Public Health*, 16(22).

Puckett, J., Byster, L., Westervelt, S., Gutierrez, R., Davis, S., Hussain, A. and Dutta, M. (2002). *Exporting harm: The high-tech trashing of Asia*. The Basel Action Network (BAN) and Silicon Valley Toxics Coalition (SVTC). Retrieved from: http://svtc.org/wp-content/uploads/technotrash.pdf.

Quan, S.X., Yan, B., Yang, F., Li, N., Xiao, X.M. and Fu, J.M. (2015). Spatial distribution of heavy metal contamination in soils near a primitive e-waste recycling site. *Environmental Science and Pollution Research*, 22(2), 1290-1298.

Robinson, B.H. (2009). E-waste: An assessment of global production and environmental impacts. *Science of The Total Environment*, 408(2), 183-191.

Salehabadi, D. (2013). *Solving the e-waste problem (StEP) green paper: Transboundary movements of discarded electrical and electronic equipment*. StEP Initiative/United Nations University. Retrieved from: https://i.unu.edu/media/unu.edu/news/34748/StEP_GP_TBM_20130325.pdf.

Saravanabhavan, G., Werry, K., Walker, M., Haines, D., Malowany, M. and Khoury, C. (2017). Human biomonitoring reference values for metals and trace elements in blood and urine derived from the Canadian Health Measures Survey 2007–2013. *International Journal of Hygiene and Environmental Health*, 220(2, Part A), 189-200.

Schenk, L., Hansson, S.O., Rudén, C. and Gilek, M. (2008). Occupational exposure limits: A comparative study. *Regulatory Toxicology and Pharmacology*, 50(2), 261-270.

Schluep, M., Hagelueken, C., Kuehr, R., Magalini, F., Maurer, C., Meskers, C., Mueller, E. and Wang, F. (2009). *Recycling from e-waste to resources*. United Nations Environment Programme. Retrieved from: http://www.unep.fr/shared/publications/pdf/DTIx1192xPA-Recycling%20from%20ewaste%20to%20Resources.pdf.

Secretariat of the Basel Convention (2002). *Amendment to the Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and their Disposal*. Retrieved from: http://www.basel.int/Countries/StatusofRatifications/BanAmendment/tabid/1344/Default.aspx.

Secretariat of the Stockholm Convention (2017). *Recommendation by the Persistent Organic Pollutants Review Committee to list short-chain chlorinated paraffins in Annex A to the Convention and draft text of the proposed amendment,* . Report No. SC-8/11: Listing of short-chain chlorinated paraffins. Châtelaine GE, Switzerland: United Nations Environment Program (UNEP). Retrieved from: http://chm.pops.int/TheConvention/POPsReviewCommittee/Recommendations/tabid/243/Default.aspx.

Secretariat of the Stockholm Convention (2020). *Dechlorane Plus and its syn- and anti-isomers: Draft risk profile* United Nations Environment Program (UNEP). Retrieved from: https://echa.europa.eu/documents/10162/df683e4a-06d5-676f-8180-106bac9bbdf4.

Secretariat of the Stockholm Convention on Persistent Organic Pollutants (2019). *Proposal to list Dechlorane Plus (CAS No. 13560-89-9) and its syn-isomer (CAS No. 135821-03-3) and anti-isomer (CAS No. 135821-74-8) in Annexes A, B and/or C to the Stockholm Convention on Persistent Organic Pollutants*. Report No. UNEP/POPS/POPRC.15/3: United Nations Environment Program (UNEP). Retrieved from: http://chm.pops.int/TheConvention/POPsReviewCommittee/Meetings/POPRC15/Overview/tabid/8052/ctl/Download/mid/22055/Default.aspx?id=31&ObjID=26981.

Sepúlveda, A., Schluep, M., Renaud, F.G., Streicher, M., Kuehr, R., Hagelüken, C. and Gerecke, A.C. (2010). A review of the environmental fate and effects of hazardous substances released from electrical and electronic equipments during recycling: Examples from China and India. *Environmental Impact Assessment Review*, 30(1), 28-41.

Sheng, P.P. and Etsell, T.H. (2007). Recovery of gold from computer circuit board scrap using aqua regia. *Waste Management & Research*, 25(4), 380-383.

Siqi, Z., Guangming, L., Wenzhi, H., Juwen, H. and Haochen, Z. (2019). Recovery methods and regulation status of waste lithium-ion batteries in China: A mini review. *Waste Management & Research*, 37(11), 1142-1152.

Song, Q. and Li, J. (2014). A systematic review of the human body burden of e-waste exposure in China. *Environment International*, 68, 82-93.

Stubbings, W.A. and Harrad, S. (2014). Extent and mechanisms of brominated flame retardant emissions from waste soft furnishings and fabrics: A critical review. *Environment International*, 71, 164-175.

Sverko, E., Tomy, G.T., Reiner, E.J., Li, Y.-F., McCarry, B.E., Arnot, J.A., Law, R.J. and Hites, R.A. (2011). Dechlorane plus and related compounds in the environment: A review. *Environmental Science & Technology*, 45(12), 5088-5098.

Syed, S. (2012). Recovery of gold from secondary sources—A review. *Hydrometallurgy*, 115-116, 30-51.

Tang, W., Cheng, J.P., Zhao, W.C. and Wang, W.H. (2015). Mercury levels and estimated total daily intakes for children and adults from an electronic waste recycling area in Taizhou, China: Key role of rice and fish consumption. *Journal of Environmental Sciences*, 34, 107-115.

Tang, Z., Huang, Q., Cheng, J., Yang, Y., Yang, J., Guo, W., Nie, Z., Zeng, N. and Jin, L. (2014). Polybrominated diphenyl ethers in soils, sediments, and human hair in a plastic waste recycling area: A neglected heavily polluted area. *Environmental Science and Technology*, 48(3), 1508-1516.

The World Bank (2020) *World Bank Open Data: Free and open access to global development data.* Retreived from https://data.worldbank.org/.

Tian, M., Chen, S.-J., Wang, J., Zheng, X.-B., Luo, X.-J. and Mai, B.-X. (2011). Brominated flame retardants in the atmosphere of e-waste and rural sites in southern China: Seasonal variation, temperature dependence, and gas-particle partitioning. *Environmental Science & Technology*, 45(20), 8819-8825.

Townsend, T.G. (2011). Environmental issues and management strategies for waste electronic and electrical equipment. *Journal of the Air & Waste Management Association*, 61(6), 587-610.

Tsydenova, O. and Bengtsson, M. (2011). Chemical hazards associated with treatment of waste electrical and electronic equipment. *Waste Management*, 31(1), 45-58.

Tzoraki, O., Zkeri, E., Lasithiotakis, M. and Sinioros, P. (2019). Trace metals' contamination in water and soils in the vicinity of a small–medium waste electrical and electronic equipment recycling plant. *Environmental Progress and Sustainable Energy*, 10.1002/ep.13343.

UN Environment (2017). *The 16 New POPs: An introduction to the chemicals added to the Stockholm Convention as Persistent Organic Pollutants by the Conference of the Parties*. Geneva, Switzerland: S. C. Secretariat. Retrieved from: http://www.pops.int/TheConvention/ThePOPs/TheNewPOPs/tabid/2511/Default.aspx.

United States Environmental Protection Agency (1991). *Risk assessment guidance for superfund: Volume I - Human health evaluation manual (Part B, development of risk-based preliminary remediation goals)*. Washington, DC. Retrieved from: https://epa-prgs.ornl.gov/radionuclides/HHEMB.pdf.

United States Environmental Protection Agency (2001). *Risk assessment guidance for superfund (RAGS), Vol III—Part A, process for conducting probabilistic risk assessment*. Washington, DC. Retrieved from: https://www.epa.gov/sites/production/files/2015-09/documents/rags3adt_complete.pdf?fbclid=IwAR2IPyJ1Hy9grAX3d7-Wkrm_iMKvVoiBKwNw4TLiURpukzeBZ-wYodQIInY.

United States Environmental Protection Agency (2017). *Exposure factors handbook chapter 5 (Update): Soil and dust ingestion.* Washington, DC. Retrieved from: https://www.epa.gov/sites/production/files/2018-01/documents/efh-chapter05_2017.pdf.

United States Environmental Protection Agency (2019a) *Regional screening levels (RSLs) - generic tables.* Retreived from https://www.epa.gov/risk/regional-screening-levels-rsls-generic-tables?fbclid=IwAR2ajG961h3zDpNyYoHFF8N_DYjFN9fr7fRRi5NN8mjYYm_86iNDuP7FOuU.

United States Environmental Protection Agency (2019b). *Regional screening levels (RSLs) - Generic tables (TR=1E-06 THQ=0.1)*. United States Environmental Protection Agency. Retrieved from: https://semspub.epa.gov/src/document/HQ/199657.

United States Environmental Protection Agency (2020) *U.S. Environmental Protection Agency*. Retreived from https://www.epa.gov/.

Vaccari, M., Vinti, G., Cesaro, A., Belgiorno, V., Salhofer, S., Dias, M.I. and Jandric, A. (2019). WEEE treatment in developing countries: Environmental pollution and health consequences—An overview. *International Journal of Environmental Research and Public Health*, 16(9).

Velis, C.A., Wilson, D.C., Rocca, O., Smith, S.R., Mavropoulos, A. and Cheeseman, C.R. (2012). An analytical framework and tool ('InteRa') for integrating the informal recycling sector in waste and resource management systems in developing countries. *Waste Management & Research*, 30(9_suppl), 43-66.

Venkatesan, A.K. and Halden, R.U. (2014). Brominated flame retardants in U.S. biosolids from the EPA national sewage sludge survey and chemical persistence in outdoor soil mesocosms. *Water Research*, 55, 133-142.

Wang, X.T., Zhang, Y., Miao, Y., Ma, L.L., Li, Y.C., Chang, Y.Y. and Wu, M.H. (2013). Short-chain chlorinated paraffins (SCCPs) in surface soil from a background area in China: occurrence, distribution, and congener profiles. *Environmental Science and Pollution Research*, 20(7), 4742-4749.

Webster, T.F., Harrad, S., Millette, J.R., Holbrook, R.D., Davis, J.M., Stapleton, H.M., Allen, J.G., McClean, M.D., Ibarra, C., Abdallah, M.A.-E. and Covaci, A. (2009). Identifying transfer mechanisms and sources of decabromodiphenyl ether (BDE 209) in indoor environments using environmental forensic microscopy. *Environmental Science & Technology*, 43(9), 3067-3072.

Wong, C.S., Duzgoren-Aydin, N.S., Aydin, A. and Wong, M.H. (2007). Evidence of excessive releases of metals from primitive e-waste processing in Guiyu, China. *Environmental Pollution*, 148(1), 62-72.

World Health Organization (2012). *Rapid risk assessment of acute public health events*. Report No. WHO/HSE/GAR/ARO/2012.1. Geneva, Switzerland: World Health Organization. Retrieved from: https://www.who.int/csr/resources/publications/HSE_GAR_ARO_2012_1/en/.

World Health Organization (2020) *World health data platform*. Retreived from https://www.who.int/data.

Wu, Q.H., Leung, J.Y.S., Du, Y.M., Kong, D.G., Shi, Y.F., Wang, Y.Q. and Xiao, T.F. (2019). Trace metals in e-waste lead to serious health risk through consumption of rice growing near an abandoned e-waste recycling site: Comparisons with PBDEs and AHFRs. *Environmental Pollution*, 247, 46-54.

Xu, X., Zeng, X., Boezen, H.M. and Huo, X. (2015). E-waste environmental contamination and harm to public health in China. *Frontiers of medicine*, 9(2), 220-228.

Zeng, X., Xu, X., Boezen, H.M., Vonk, J.M., Wu, W. and Huo, X. (2017). Decreased lung function with mediation of blood parameters linked to e-waste lead and cadmium exposure in preschool children. *Environmental Pollution*, 230, 838-848.

Zeng, X., Xu, X., Zheng, X., Reponen, T., Chen, A. and Huo, X. (2016). Heavy metals in PM2.5 and in blood, and children's respiratory symptoms and asthma from an e-waste recycling area. *Environmental Pollution*, 210, 346-353.

Zhang, T., Ruan, J.J., Zhang, B., Lu, S.Y., Gao, C.Z., Huang, L.F., Bai, X.Y., Xie, L., Gui, M.W. and Qiu, R.L. (2019). Heavy metals in human urine, foods and drinking water from an e-waste dismantling area: Identification of exposure sources and metal-induced health risk. *Ecotoxicology and Environmental Safety*, 169, 707-713.

Zhang, Y., Xu, X., Sun, D., Cao, J., Zhang, Y. and Huo, X. (2017). Alteration of the number and percentage of innate immune cells in preschool children from an e-waste recycling area. *Ecotoxicology and Environmental Safety*, 145, 615-622.

Zheng, X.B., Huo, X., Zhang, Y., Wang, Q.H., Zhang, Y.L. and Xu, X.J. (2019). Cardiovascular endothelial inflammation by chronic coexposure to lead (Pb) and polycyclic aromatic hydrocarbons from preschool children in an e-waste recycling area. *Environmental Pollution*, 246, 587-596.