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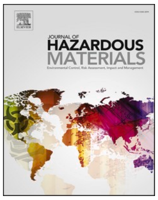

Review

Occurrence and fate of polycyclic aromatic hydrocarbons from electronic waste dismantling activities: A critical review from environmental pollution to human health

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ABSTRACT

Electronic waste (e-waste) is one of the fastest-growing solid wastes and has become an urgent issue due to the potential adverse consequences of exposure to emitted toxic pollutants, especially for these occupational exposed workers and local residents. In this review, the environmental occurrences, emission characteristics, sources, and possible adverse effects of polycyclic aromatic hydrocarbons (PAHs) emitted from primitive e-waste dismantling activities are summarized. In general, the atmospheric levels of PAHs at typical e-waste sites, e.g., in Guiyu, China, have substantially decreased by more than an order of magnitude compared with levels a decade ago. The PAH concentrations in soil from old e-waste sites in China are also generally lower than those at newly emerged e-waste sites in India, Pakistan and Ghana. However, elevated concentrations of PAHs have been reported in human milk, hair and urine from the populations near these e-waste sites. Source apportionment both from bench-scale studies to field observations has demonstrated that the pyrolysis and combustion processing of electronic circuit board are mainly responsible for the emissions of various PAHs. In addition, some specific PAHs and their derivatives, such as triphenylbenzene, halogenated and oxygenated PAHs, have frequently been identified and could be considered as indicators in routine analysis in addition to the 16 U.S. EPA priority PAHs currently used.

1. Introduction

Amounts of end-of-life electrical and electronic products (known as electronic waste or e-waste), such as mobile phones, computers, fax machines, refrigerators and televisions, have increased substantially over recent years and have become a rapidly growing global problem (Zhang et al., 2012; Perkins et al., 2014). E-waste contains reusable electronic components and precious metals that have an economic value when it was recycled. However, most e-waste is improperly processed by an unregulated profit-driven informal sector, which could pose a significant threat to e-waste dismantlers through exposure to toxic pollutants (Awasthi et al., 2016). Inappropriate e-waste dismantling practices, e.g., involving open burning, acid baths and heating of circuit boards, have caused the emission and release of hazardous pollutants, including heavy metals, persistent organic pollutants (POPs), and volatile organic pollutants (VOCs) (Wong et al., 2007; Zhang et al., 2011). During the past two decades, numerous researches have been carried out on the environmental occurrence and fate, transformation and potential adverse health consequences of these hazardous chemicals from e-waste sites (Chen et al., 2011; Grant et al., 2013; Xu et al., 2015; Lu et al., 2016). Their results have indicated that dismantling employees as well as residents near the e-waste sites are highly exposed through dermal contact with these hazardous materials, inhalation of toxic fumes and ingestion and accumulation of toxic chemicals from contaminated soil, water and food (Zhang et al., 2012; Song and Li, 2014). Attentions have

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Nomenclature

1-Cl-Pyr	1-Chloropyrene
1-MeNap	1-Methylnaphthalene
1-OH-Nap	1-Hydroxynaphthalene
1-OH-Phe	1-Hydroxyphenanthrene
1-OH-Pyr	1-Hydroxypyrene
1,2,4-TPB	1,2,4-Triphenylbenzene
1,3,5-TPB	1,3,5-Triphenylbenzene
2-MeNap	2-Methylnaphthalene
2-OH-Flu	2-Hydroxyfluorene
2-OH-Nap	2-Hydroxynaphthalene
2-OH-Phe	2-hydroxyphenanthrene
3-Cl-Fluo	3-Chlorofluoranthene
3-OH-Phe	3-Hydroxynaphthalene
3 + 2 N-Fluo	3 + 2-Nitrofluoranthene
4-OH-Phe	4-Hydroxyphenanthrene
5 N-Ace	5-Nitroacenaphthene
6-Cl-BaP	6-Chlorobenzo(a)pyrene
7 N-BaA	7-Nitrobenzo[*a*]anthracene
8-OHdG	8-Hydroxy-2′-deoxyguanosine
9-Br-Phe	9-Bromophenanthrene
9-Cl-Phe	9-Chlorophenanthrene
9FLO	9-Fluorenone
9 N-Ant	9-Nitroanthracene
9 N-Phe	9-Nitrophenanthrene
9-OH-Flu	9-Hydroxyfluorene
9-OH-Phe	9-Hydroxyphenanthrene
9,10-Br_2-Ant	9,10-Dibromoanthracene
$\sum_{7c}$-PAH	Sum of seven carcinogenic PAHs
ABS	Acrylonitrile butadiene styrene
Ace	Acenaphthene
Acy	Acenaphthylene
AhR	Aryl hydrocarbon receptor
Ant	Anthracene
ATQ	9,10-Anthraquinone
B[*a*]A	Benzo[*a*]anthracene
B[*a*]P	Benzo[a]pyrene
B[*b*]F	Benzo[*b*]fluoranthene
B[ghi]P	Benzo[*g*,*h*,*i*]perylene
B[*k*]F	Benzo[*k*]fluoranthene
Br-PAHs	Brominated polycyclic aromatic hydrocarbons
Chr	Chrysene
Cl-PAHs	Chlorinated polycyclic aromatic hydrocarbons
D[*a*,*h*]A	Dibenzo[*a*,*h*]anthracene
DBaeP	Dibenzo[*a*,*e*]pyrene
DBahP	Dibenzo[*a*,*h*]pyrene
DBaiP	Dibenzo[*a*,*i*]pyrene
DBalP	Dibenzo[*a*,*l*]pyrene
DBP	Dibenzopyrene
dw	dry weight
EDS	Electronic waste dismantling site
EFs	Emission factors
ESS	Electronic waste storage or dump site
E-waste	Electronic waste
Flu	Fluorene
Fluo	Fluoranthene
FPUF	Flexible polyurethane foam
HDPE	High density polyethylene
HIPS	High-impact polystyrene
HMW	High molecular weight
HPAHs	Halogenated polycyclic aromatic hydrocarbons
IcdP	Indeno[1,2,3,-cd]pyrene
IGF	Insulin-like growth factor
IGFBP	IGF-binding protein
LMW	Low molecular weight
LDPE	Low density polyethylene
MDA	Malondialdehyde
MeChy	Methyl chrysene
MeNap	Methyl naphthalene
MePAHs	Methylated polycyclic aromatic hydrocarbons
MePhe	Methyl phenanthrene
MPV	Mean platelet volume
MPVP	Mean platelet volume to platelet count
MW302 PAHs	PAHs with a molecular weight greater than 300
Nap	Naphthalene
NLRP3	NLR family pyrin domain containing 3
NPAHs	Nitrated polycyclic aromatic hydrocarbons
OBS	Open burning site
OH-PAHs	Monohydroxy polycyclic aromatic hydrocarbons
OPAHs	Oxygenated polycyclic aromatic hydrocarbons
PAHs	Polycyclic aromatic hydrocarbons
PBBs	Polybrominated biphenyls
PBDEs	Polybrominated diphenyl ethers
PCBs	Polychlorinated biphenyls
PCDD/Fs	Polychlorinated dioxins/dibenzofurans
PCL	Polycaprolactone
PDW	Platelet distribution width
PE	Polyethylene
PET	Polyethylene terephthalate
Phe	Phenanthrene
P-LCR	Platelet-large cell ratio
POPs	Persistent organic pollutants
PP	Polypropylene
PS	Polystyrene
PVC	Polyvinyl chloride
Pyr	Pyrene
SRM	Standard Reference Material
TPB	Triphenylbenzene
U.S. EPA	US Environmental Protection Agency
VOCs	Volatile organic pollutants
WPCBs	Waste printed circuit boards

been paid to investigate the emissions and characteristics of pollutants particularly in China, where most e-waste from all parts of the world is recycled and dismantled (Zhang et al., 2012). Owing to the potential detriment of e-waste to both humans and ecosystems, many countries have issued legislation to regulate informal e-waste dismantling activities. For example, a ban on imports of e-waste has been officially implemented in China since 2018, and the illegal treatment of domestic e-waste has also been prohibited by local government (Lin et al., 2020a). However, numerous e-waste dismantling sites have emerged in other developing countries, such as the Philippines, India, Vietnam and Ghana, still posing a potential threat to local populations and ecosystems (Daso et al., 2016; Tue et al., 2017; Chakraborty et al., 2019). Therefore, it is critical to estimate the characteristics of PAHs emitted by extensive e-waste dismantling activities, which can serve as a scientific basis for implementing pollution regulations and emission controls at emerging e-waste sites in developing countries, and provide important information for the assessment of the environmental and health risks of globally increasing e-waste trends.

Among all the organic pollutants emitted from e-waste dismantling activities, polycyclic aromatic hydrocarbons (PAHs) are of great concern owing to their mutagenic and carcinogenic potencies to humans (Chen et al., 2011; Guo et al., 2012). Numerous researches are concerning about the distribution and fate of 16 priority PAHs as defined by the US Environmental Protection Agency (U.S.EPA) (Fig. S1). High levels of 16 priority PAHs in multi-environmental matrices from e-waste dismantling and recycling areas have been extensively reported, suggesting potential threats to the health of occupationally exposed workers (Wong et al., 2007; Wang et al., 2012a; Lu et al., 2016). In general, the emissions of PAHs are mainly due to the pyrolysis of various plastic polymers in e-waste, including baking waste printed circuit boards (WPCBs) directly for recycling the reusable components and open burning of plastic-insulated wires to extract precious metals (Font et al., 2011; Cai et al., 2018). However, the characteristics of PAHs release from e-waste dismantling have not been thoroughly investigated. The profile of the 16 PAHs from e-waste dismantling areas is similar to those from other sources, such as coal combustion, and diesel and gasoline engine exhausts, complicating the source apportionment of PAHs (Gu et al., 2010; Zhang et al., 2011). There is also a lack of specific indicator or tracer to differentiate e-waste sources from other emissions. Some studies have proposed specific tracers, e.g., 1,3,5-triphenylbenzene (1,3,5-TPB) released from the burning of plastics (Gu et al., 2010; Simoneit, 2015) can be used as a tracer in source apportionment, however, need more field observations to verify. Furthermore, during the heating of WPCBs and plastics, various derivatives of PAHs are also generated, including brominated PAHs (Br-PAHs), chlorinated PAHs (Cl-PAHs), alkylated PAHs, nitrated PAHs (NPAHs) and oxygenated PAHs (OPAHs) (Ma et al., 2009; Wei et al., 2012; Tue et al., 2017; Wang et al., 2016; Huang et al., 2019; Tang et al., 2020). Some of these derivatives could be used as indicators to track emissions from e-waste sources, and therefore could be included in PAH monitoring soon.

At present, several review articles have summarized the concentration levels, composition profiles and health risks of e-waste various pollutants. For instance, Wong et al. (2007) surveyed the levels of heavy metals and several POPs, including PAHs from different environmental matrices in Guiyu, and concluded that the environment was seriously polluted by the harmful substances from informal e-waste dismantling processes. Song and Li (2014) reviewed the exposure routes and body burden of heavy metals, polychlorinated dioxins/dibenzofurans (PCDD/Fs), polychlorinated biphenyls (PCBs), polybrominated diphenyl ethers (PBDEs), and polybrominated biphenyls (PBBs) emitted from e-waste pollution in China, and found that residents near the e-waste recycling sites suffered from a high daily intake of e-waste pollutants, with children and neonates being one of the most vulnerable groups. However, there has not been a systematic review of the occurrence and fate of PAHs from e-waste recycling activities yet, especially concerning environmental pollution and human health. Thus, the aim of this review was to provide systematical information on PAHs generated from extensive e-waste recycling activities with different characteristics, emission sources, formation mechanisms, human body burdens and potential health consequences. For this purpose, the electronic databases including the Web of Science, PubMed, and Google Scholar were searched to identify potential articles about the occurrence and fate of polycyclic aromatic hydrocarbons from e-waste dismantling activities. The search terms were defined as "*polycyclic aromatic hydrocarbons*" and "*electronic waste*", and our search was further expanded by considering the references of the obtained articles. Only studies published in English before July 2021 were eligible for inclusion.

2. PAHs and derivatives in the environment and biotas at e-waste sites

2.1. PAHs in the environment and biotas at e-waste sites

2.1.1. PAHs in the various environmental matrixes

As shown in Table 1, PAHs in air particulate samples have frequently been investigated at three typical e-waste recycling areas in China, i.e., Taizhou, Guiyu, and Qingyuan. Significantly higher levels of PAHs were observed in particulate matter compared with surrounding reference sites. For example, PAH concentrations at e-waste sites were about two times higher than the reference site in both summer and winter seasons ($p < 0.01$ in paired *t*-test) (Gu et al., 2010), indicating that the extensive e-waste recycling activity resulted in severe air pollution. As reported by Wong et al. (2007), the average level of the most toxic benzo[*a*]pyrene (B[*a*]P) at Guiyu e-waste site was 2–6 times higher compared with other Asian cities. However, no significant difference was detected between the levels of 16 priority PAHs in Qingyuan e-waste dismantling area (median 18.3 ng/m^3) and at the heavily polluted metropolis Guangzhou site (median 14.3 ng/m^3) (Chen et al., 2016). In addition to e-waste dismantling activities, emissions of 16 priority PAHs may be related to many other sources, such as vehicle emissions, wood and coal combustion. In contrast, levels of molecular weight greater than 300 (MW 302 PAHs) in e-waste area (median 3.05 ng/m^3) were two times higher

Table 1
Atmospheric concentrations and pollution profiles of PAHs (ng/m^3) from different e-waste sites.

Samples	E-waste sites		Surrounding sites		Refs
	Concentration Median (range)	Composition profiles	Concentration Median (range)	Composition profiles	
Guiyu PM$_{2.5}$ (2004)	102 ± 63.6 (22.7–263)[a]	B[bk]F > IcdP > B[ghi]P > B[*a*]P > Chr > Pyr[b]	na[c]	na[c]	(Deng et al., 2006)
Taizhou PM$_{2.5}$ (2005–2006)	248.5 ± 25.36[a]	B[bk]F > Chr > B[*a*]A > B[*a*]P ≈ IcdP ≈ B[ghi]P	129.4 ± 24.94[a]	B[bk]F > Chr > B[ghi]P ≈ IcdP > B[*a*]P ≈ B[*a*]A	(Gu et al., 2010)
Guiyu Gaseous phase and TSP (2007)	(313–1041)	Phe > Flu > Pyr	50.1	Flu > Phe > Pyr	(Zhang et al., 2011)
Qingyuan TSP (2009–2010)	(7.38–87.9)	B[ghi]P > IcdP > B[*b*]F > B[*a*]P > Fluo > B[*k*]F	(16.7–113)[d]	B[ghi]P > IcdP > B[*b*]F > B[*a*]P > B[*k*]F > Fluo	(Wei et al., 2012)
Qingyuan TSP (2010–2011)	18.3 (4.4–89.7)	B[*b*]F > B[ghi]P > IcdP > B[*a*]P > Fluo > B[*k*]F	14.3 (3.95–61.4)[d]	B[*b*]F > B[ghi]P > IcdP > Fluo > B[*a*]P > B[*k*]F	(Chen et al., 2016)
Qingyuan PM (2012)	(15.1–17.7)[e]	B[ghi]P > Phe > IcdP > B[*b*]F > B[*k*]F ≈ Chr ≈ Fluo	(5.8–6.7)[d]	B[ghi]P > IcdP > Phe ≈ Fluo > B[*b*]F ≈ B[*k*]F ≈ Chr	(Luo et al., 2015)
Guiyu PM$_{2.5}$ (2015–2016)	5.7 (2.1–45.0)	B[*b*]F > B[*k*]F > IcdP > B[ghi]P	5.7 (14.0–45.7)	B[*b*]F > B[*k*]F > IcdP > B[ghi]P	(Chen et al., 2019)
Guiyu Gaseous phase (2015–2016)	62.1 ± 9.4[a] (37.7–109)	Phe > Pyr	(12.3–36.4)	Phe > Pyr	(Chen et al., 2019)

[a] mean concentration;
[b] sum of concentration of B[b]F and B[k]F;
[c] not available;
[d] Urban sites;
[e] Average concentrations.

than that at the urban site (median 1.42 ng/m^3) ($p < 0.001$), suggesting that e-waste dismantling may be a special source of these MW 302 PAHs (Chen et al., 2016).

Besides MW 302 PAHs, several works have reported specific isomers other than the 16 U.S. EPA priority PAHs. For example, 1,3,5-TPB (Fig. 1), an atmospheric product originating from burning polyethylene (PE) plastics and landfill wastes, has frequently been reported in particulate samples at e-waste sites (Gu et al., 2010; Simoneit, 2015). Concentrations of atmospheric TPB from an e-waste site (Qingyuan) during different seasons (0.36–51.6 ng/m^3 in winter, 0.16–49.6 ng/m^3 in summer) have been shown to be about 3–10 times of those at a reference site (Wei et al., 2012) and more than 20 times of those found at Guangzhou (n.d.–0.27 ng/m^3) (Bi et al., 2002). In contrast, the concentrations of 16 priority PAHs from e-waste sites were generally no more than two times higher than that at reference sites (Wei et al., 2012; Chen et al., 2016). Gu et al. (2010) showed that TPB was exclusively detected in particles from the burning of hard plastic blocks but not from burning wires/cables. Thus, TPB might be released during the informal pyrolysis and combustion of WPCBs and could be used as a specific tracer of illegal e-waste dismantling activities. However, more research is needed to verify its specificity in source apportionment (Simoneit et al., 2005; Gu et al., 2010; Simoneit, 2015).

Seasonal variations in atmospheric concentrations of PAHs from e-waste areas were also investigated (Gu et al., 2010; Wei et al., 2011; Chen et al., 2016, 2019). In general, the main e-waste areas studied in China were located in a subtropical zone, and most of the PAH isomers exhibited relatively lower levels in the summer and autumn, and higher levels in spring and winter (Chen et al., 2016, 2019), mainly due to the Asian monsoon system. In the autumn and summer, pollutants in air are often diluted and diffused by clean air ventilation from the ocean. In the winter and spring, when the southwesterly monsoon prevails, “dirty” air masses from coal combustion regions of northern China are transported to these e-waste areas. However, the emissions of TPB were not found to be influenced by seasonal variations (Wei et al., 2012). This suggests that atmospheric TPB mainly originates from burning plastics, which is a prevalent process in routine recycling activities and is not influenced by seasonal variation. Thus, atmospheric TPB could be used as a specific tracer for e-waste source apportionment and should be listed as an e-waste priority PAH.

In addition, diurnal variations of PAH levels from e-waste sites were also studied (Wei et al., 2011, 2012; Zhang et al., 2011). Significantly higher concentrations of high molecular weight (HMW) PAHs and MW302 PAHs were found at night than those during daytime at an e-waste site, indicating that source characterization and atmospheric stability may be significant factors controlling the short-term variability of PAH isomers (Wei et al., 2011, 2012). Compared with daytime, the lower atmospheric mixing layer at night may have a concentrating effect on PAH isomers, whereas combustion of honeycomb coals, which are used to bake electronic boards, changes from burning to smoldering and could cause the continuous emission of hazardous PAHs. However, higher daytime concentration of PAHs was reported in the waste incineration field of Guiyu owing to the smoldering combustion of unwanted plastic insulating layers and municipal solid waste (Zhang et al., 2011).

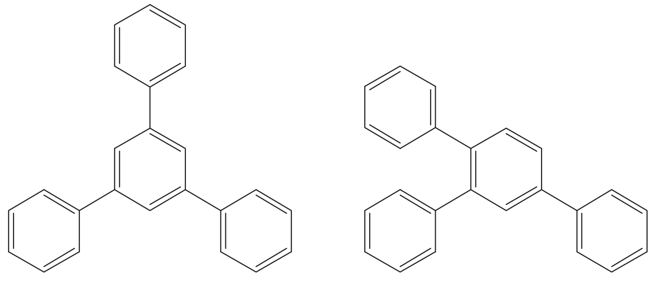

Fig. 1. Chemical structures of potential PAH tracers 1,3,5-triphenylbenzene and 1,2,4-triphenylbenzene.

Furthermore, it has been shown that PAH concentrations in atmospheric particles from typical e-waste recycling sites have sharply declined in recent times (Table 1). The median concentration of PAHs at Guiyu were decreased by more than an order of magnitude in 2016 compared with those more than a decade ago. This trend may vary with different sampling sites and sampling times. However, government efforts to regulate e-waste recycling activities may also have resulted in a reduction of hazardous chemical emissions. A declining trend of PAH concentrations in rice and rice hulls sampled from a typical e-waste dismantling site has also been reported (Liu et al., 2014). The decreasing trend in atmospheric concentrations of PAHs from e-waste recycling areas is consistent with the decreasing trend in global atmospheric emissions of PAHs (Shen et al., 2013).

The atmospheric occurrence of PAHs in air particles from different e-waste sites exhibits similar compositional patterns (Table 1). Four- to six-ring HMW PAH homologs contributed more than 80% of the total PAH levels in most studies. Benzo[*k*]fluoranthene (B[*k*]F), Benzo[*b*]fluoranthene (B[*b*]F), Indeno[1,2,3,-cd]pyrene (IcdP), and Benzo[*g*,*h*,*i*]perylene (B[ghi]P)were found to be the predominant PAH isomers in atmospheric particles (Deng et al., 2006; Gu et al., 2010; Chen et al., 2016). However, two studies indicated that volatile two- to three-ring PAHs were the richest isomers from the e-waste sites, especially in the gaseous phase, and they contributed a large portion of the total PAH values (Zhang et al., 2011; Chen et al., 2019). Chen et al. (2019) found that these gaseous phase PAHs accounted for 80.1–91.3% of the total PAHs from an e-waste recycling site. In different e-waste recycling workshops, fluorene (Flu), acenaphthene (Ace), anthracene (Ant), and phenanthrene (Phe) were observed almost exclusively in the gaseous phase (92–98%), while five- and six-ring PAHs (B[*k*]F, B[*b*]F, B[*a*]P, Dibenzo[*a*,*h*]anthracene (D[*a*,*h*]A), IcdP, and B[ghi]P) were primarily associated with the particulate phase (>98%) (Zhang et al., 2011). In addition, four carcinogenic dibenzopyrene (DBP) isomers, including dibenzo[*a*,*e*]pyrene (DBaeP), dibenzo[*a*,*l*]pyrene (DBalP), dibenzo[*a*,*h*]pyrene (DBahP), and dibenzo[*a*,*i*]pyrene (DBaiP), were found from e-waste sites. Their distribution was close to that of coal-tar extract (Standard Reference Material (SRM) 1597) and air particles from the urban city of Beijing but distinct from that of diesel particles (SRM 2975, SRM 1650). The results indicated that coal combustion might have contributed significantly to the MW302 isomers (Wei et al., 2011). It is consistent with the fact that coal combustion is widely used in illegal e-waste dismantling activities, e.g., baking electronic boards on open fires to more easily remove different parts.

Most isomers of PAHs have been widely detected in sediment and soil from e-waste dismantling sites, with total concentrations ranging from 25 to 37,000 ng/g dry weight (dw) (Table 2). The highest total concentrations of PAHs 37,000 ng/g dw were detected in soil samples collected at a typical e-waste dismantling workshop of northern Vietnam, indicating the presence of localized sources (Hoa et al., 2020). The median levels of total PAHs in soil samples from old e-waste sites in China are generally lower than from newly emerged sites in India, Pakistan and Ghana (Daso et al., 2016; Jiang et al., 2017; Chakraborty et al., 2019), indicating that the prevalence of e-waste dismantling causes widespread PAH contamination. In most studies, the sampling site of the e-waste area was divided according to different dismantling process, e.g., open burning site (OBS), e-waste dismantling site (EDS), e-waste storage or dump site (ESS). The concentrations of PAHs from different sites were found to be in the following descending order: OBS > EDS > ESS (Fig. 2a). The burning of plastics and wires/cables during e-waste recycling processes may be a major contributor to the high concentrations of PAHs in the soil samples. Huang et al. (2014) reported that soils below 20 cm deep (60–80 cm deep) in former open burning areas were polluted, even though burning e-waste had been banned for

Table 2
Concentrations and profiles of PAHs in soil and sediment (ng/g dw) from different e-waste sites.

Samples	Concentration Median (range)	Composition Profiles	Refs
Guiyu, Soil (2004)	389 (45–3206)	Phe > Nap > Chr > Pyr > B[bk]F > B[ghi]P	(Yu et al., 2006)
Taizhou, Soil (2006)	640 (330–20,000)	Fluo > B[*b*]F ≈ BaA ≈ Pyr ≈ Chr	(Shen et al., 2009)
Taizhou, Soil (2008)	329 (39–708)[a]	Fluo > B[*b*]F > Pyr > Phe > Ant > Chr	(Tang et al., 2010)
Hong Kong, Soil (2008)	1008 (107–2300)	B[*b*]F > Fluo > Pyr > BaP ≈ IcdP ≈ B[ghi]P	(Lopez et al., 2011)
Guiyu, Soil (2009)	364 (174–510)[a]	Phe > Fluo	(Xu et al., 2016)
Qingyuan, Soil (2011)	514 (25–4300)[a]	Nap > Phe > Fluo > Pyr > Chr > B[*b*]F	(Huang et al., 2014)
Taizhou, Soil (2011)	480 (191–1922)	IcdP > B[ghi]P > Phe > B[bk]F > Fluo > D[*a*,*h*]A	(He et al., 2019)
Guiyu, Soil and Ash (2012)	740 (295–10,084)	Phe > Fluo > Pyr > Nap > Chr >B[*a*]A	(Gao et al., 2015)
Guiyu, Soil (2012)	(72–2506)	Nap > Phe > Fluo ≈ Pyr ≈ B[*b*]F ≈ B[*a*]A	(Luo et al., 2017)
Taizhou, Soil (2016)	356 (59–1332)	B[bk]F > IcdP > Phe > B[ghi]P > Fluo > Pyr	(He et al., 2019)
Taizhou, Soil (n.a.)	327 (90–1054)	Phe > B[*b*]F > B[*k*]F > Ant > Fluo > Pyr	(Gu et al., 2019)
Pakistan, Soil (2012)	2350 (2180–2940)	n.a.	(Jiang et al., 2017)
India, Soil (2014)	866 (255–3328)	Nap > Phe > Ant > Pyr > Fluo > B[*a*]A	(Chakraborty et al., 2019)
Ghana, Soil (2015)	3000 (2679–4822)	Nap ≈ Phe ≈ Fluo > Pyr > Chr > B[*a*]A	(Daso et al., 2016)
Vietnam, Soil (2012)	2200 (720–37,000)	Fluo > Phe > Pyr ≈ B[bk]F > Chr	(Hoa et al., 2020)
Guiyu, Sediment (2012)	479 (67–4766)	Fluo > B[*a*]A ≈ Chr > B[*b*]F > Flu > B[*k*]F	(Gao et al., 2015)
Taizhou, Sediment (2007)	3340 (809–7880)	Phe > Ant > Pyr > B[*b*]F > Fluo > B[*k*]F	(Chen et al., 2010)
Guiyu, Sediment (2009)	985 (181–3034)[a]	Phe > Fluo ≈ Pyr ≈ Chr	(Xu et al., 2016)
Guiyu, Sediment (2012)	(802–4902)	Nap > Phe > Fluo > Pyr	(Liu et al., 2018)
Vietnam, Soil (2012)	650 (340–2100)	Pyr > B[bk]F > Fluo > Chr > Phe	(Hoa et al., 2020)

[a] mean concentration

several years before sampling. Although a sharp decrease in the concentrations of PAHs was found in the atmosphere, levels of PAHs in the surface soil samples did not show a clear trend despite a strict environmental policy. It has also been shown that the mean concentration of PAHs in Taizhou decreased from 590 ng/g in 2011 to 407 ng/g in 2016, and the percentage of non-contaminated soil increased from 2.3% in 2011 to 14.0% in 2016 (He et al., 2019).

The compositional profiles of PAHs vary significantly among different e-waste sites. As shown in Fig. 2b, relatively higher contributions of low molecular weight (LMW) PAHs were found at highly contaminated OBS compared with EDS and ESS. Especially in newly emerged e-waste dismantling sites, such as at Agbogbloshie, Ghana, one of the largest e-waste dump/dismantling areas in Africa, PAH compositional profiles were dominated by two- to three-ring PAHs, with a percentage of 60% in open burning sites (Daso et al., 2016). A predominance of relatively volatile PAHs (naphthalene (Nap) to Phe) was also observed in most soil sampled from e-waste sites in India, with contributions of more than 50%, reflecting a relatively fresh source, which may be from the incomplete combustion during e-waste recycling activities (Chakraborty et al., 2019). At the same sites, D[*a*,*h*]A was nondetectable in all the soil samples, while HMW isomers B[*k*]F, B[*b*]F, B[*a*]P, B[ghi]P and IcdP were also lower than the detection limits in soils from an e-waste shredding site (Chakraborty et al., 2019). These results indicate that LMW PAHs dominate in heavily PAH-impacted sites. On the other hand, several studies have reported a predominance of HMW isomers in Taizhou and Guiyu, China (He et al., 2019; Gao et al., 2015).

In general, the occurrences of PAHs have been widely reported in air, soil and sediment samples collected from typical e-waste sites. Since the characteristics of PAHs were highly related to the sampling sites and sampling times, studies with similar sampling time at the same site were selected to discuss the variations of PAHs profiles between different environmental matrices. HMW PAHs with five- to six-ring homologs contributed more than 70% of the total PAH levels in TSP and $PM_{2.5}$, while LMW PAHs such as Nap, Phe, Fluo were found to be the predominated isomers in soil (Deng et al., 2006; Yu et al., 2006). However, some studies also indicated that the more volatile LMW PAHs were mainly distributed in gaseous phase (Zhang et al., 2011; Chen et al., 2019), which was similar to the distribution in soil and sediment. This might be due to the fact that soil contamination was mainly derived from atmospheric wet and dry deposition. For these sampling sites that are being polluted by e-waste dismantling activities, PAHs have not reached an equilibrium state among different environmental matrices. The loss of relatively volatile LMW PAHs from once contaminated soils could thus cause a proportional increase of HMW PAHs. In addition, similar composition homologues of PAHs have been found at different depths of soil profiles, indicating that PAHs can be transported together down through soil profile (Huang et al., 2014). More systematic research is needed to clarify the key factors and mechanisms affecting their environmental distribution.

2.1.2. PAHs in biotas as well as in human body

Levels and profiles of 16 priority PAHs in plants from e-waste recycling sites in Nigeria and China have been measured previously. The highest concentration of total PAHs (up to 14.95 μg/g dw) was reported in plants from an e-waste site in Nigeria (Alabi et al., 2012). In China, the highest concentration was also found in plants from Guiyu (2.53–4.38 μg/g dw) (Alabi et al., 2012), which was almost one order of magnitude higher than those at two other major e-waste dismantling areas in China (Qingyuan: 199–2420 ng/g; Taizhou: 40.8–412 ng/g) (Wang et al., 2012b; Liu et al., 2014). Moreover, plants from Guiyu possessed higher percentages of carcinogenic PAHs compared to that from other e-waste recycling areas (Wang et al., 2012b). Wang et al. (2017) found that concentrations of PAHs in plant shoots and roots from an e-waste dismantling area in Southern China ranged from 96 to 388 ng/g and 143 to 605 ng/g, respectively. Similarly, PAH concentrations were in the range of 40.8–138 ng/g in rice and 67.7–412 ng/g in rice hulls from an e-waste recycling site in South China (Liu et al., 2014). In all these studies, Phe dominated the concentration profile in all plant samples, except in the study of Wang et al. (2012b), where fluoranthene (Fluo), benzo[*a*]anthracene (B[*a*]A), and pyrene (Pyr) were the predominant isomers in plants. In Wang et al. (2012b)'s study, the PAH profile in plants was different from that in soil, demonstrating that the uptake of PAHs by plants through roots or foliage may be species- as well as tissues-dependent. In addition, PAHs have been measured in marine organisms from a bay or harbor near to the e-waste recycling area. The PAH concentrations in ten different mollusk samples were in the range of 56.4–432 ng/g dw, with predominant congeners of Nap, Phe, Fluo, Pyr and IcdP depending on the species (Liu et al., 2014). Concentrations of PAHs in seven species in marine fish were also measured as a range of 388.5–5640 ng/g (wet weight), with the most abundant congeners being three-ring PAHs (Shi et al., 2016). Similar to plants, PAH profiles in marine organisms seem to vary from species to species, which might also result in a distinct accumulation pattern for different species.

Blood, breast milk, urine and hair are the most commonly used

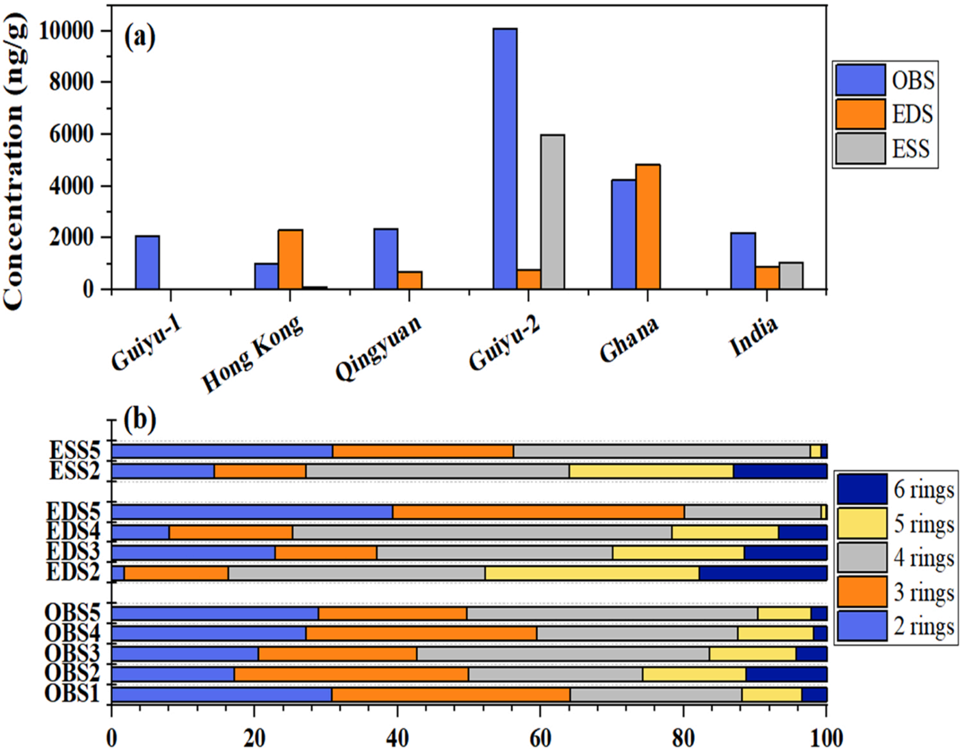

Fig. 2. Concentrations (a) and composition profiles (b) of PAHs in soils from three typical sampling sites: open burning site (OBS), e-waste dismantling site (EDS), e-waste storage or dump site (ESS). The references are cited as follow, Guiyu-1 from Yu et al. (2006); Hong Kong from Lopez et al. (2011); Qingyuan from Huang et al. (2014); Guiyu-2 from Gao et al. (2015); Ghana from Daso et al. (2016); India from Chakraborty et al. (2019).

biological matrices for monitoring organic compounds in exposure and environmental epidemiology studies for human being. However, few works have concentrated on blood levels of PAHs related to e-waste sites, mainly due to their short half-lives in the human body. After entering human body, PAHs are rapidly metabolized by cytochrome P450 enzymes into their hydroxylated metabolites (OH-PAHs), followed by the formation of more water-soluble glucuronidated/sulfated conjugates, thus promote their excretion either through urine or bile (Li et al., 2010). Therefore, the estimation of the body burden of PAHs was commonly achieved by monitoring the urinary concentrations of OH-PAHs. Xu et al. (2013, 2015) reported the concentrations of 16 PAHs in umbilical cord blood samples with a median concentration of 14.43 ppb and child blood samples with a median concentration of 68.53 μg/L from Guiyu, which were significantly higher than those in the reference group. Moreover, the concentrations of individual PAHs were higher in children whose father worked in e-waste dismantling sites than those whose father did not, especially for carcinogenic PAH congeners. The distribution profiles of PAHs in umbilical cord blood samples and child blood samples were similar, with three-, five-, and six-ring PAHs being the richest species. Guo et al. (2012) determined the body burden of seven carcinogenic PAHs in neonates from Guiyu and at a reference site using umbilical cord blood. The median Σ_{7c}-PAH (sum of seven carcinogenic PAHs) concentration was obtained as 108.05 and 79.36 ppb for Guiyu and the reference site, respectively. The predominant compound among the seven carcinogenic PAHs was B[*k*]F. However, besides residence time, pregnant women spend longer time cooking food in the kitchen during the gestation period was also identified as a significant factor contributing to the Σ_{7c}-PAH concentration.

Only one study has reported the occurrence of PAHs in breast milk of the women at an e-waste dismantling site, and only 16 priority PAHs and two methylnaphthalene (MeNap) isomers were reported. The concentration of $\sum_{18}$PAH in breast milk ranged from 0.30 to 15,800 ng/g lipid weight, which was one to two orders of magnitudes higher than those from a residential area (<LOD to 687 ng/g lipid weight) (Asamoah et al., 2019). Nap was the dominant isomer in the milk samples, in contrast to the results from blood samples. In addition, the most carcinogenic PAHs, e.g., Chrysene (Chr), B[*k*]P, B[*a*]F and B[ghi]P, in milk samples were below the limit of detection in the residential area but were detected in e-waste dismantling area (Asamoah et al., 2019), which contrasts to the results from umbilical cord blood samples (Guo et al., 2012).

Compared with blood and breast milk, urine is an effective biological fluid to evaluate human exposure to PAHs in environment. Instead of PAHs, OH-PAHs are a kind of target compounds during urine analysis. Many studies have measured OH-PAHs in human urine sampled from e-waste recycling sites. Generally, 2-hydroxynaphthalene (2-OH-Nap), 1-hydroxynaphthalene (1-OH-Nap), 4-hydroxyphenanthrene (4-OH-Phe), 3-hydroxyphenanthrene (3-OH-Phe), 2-hydroxynaphthalene (2-OH-Phe), 1-hydroxyphenanthrene (1-OH-Phe), 9-hydroxyphenanthrene (9-OH-Phe), 9-hydroxyfluorene (9-OH-Flu), 2-hydroxyfluorene (2-OH-Flu), and 1-hydroxypyrene (1-OH-Pyr) are believed as the ten most widely determined OH-PAHs. These ten OH-PAHs have been frequently identified in the urine of preschool children (Dai et al., 2019; Zheng et al., 2019), pregnant (Huang et al., 2020) and lactating women (Huo et al., 2019) from e-waste dismantling area in Guiyu, and their concentrations are significantly higher than those of reference groups. Moreover, 2-OH-Nap and 1-OH-Nap are often the predominant congeners presented in urine samples. Lu et al. (2016) measured the concentrations of ten OH-PAHs in urine samples from residents near an e-waste recycling site in Qingyuan. The results found that occupationally exposed workers exhibited significantly higher concentrations (36.6 μg/g creatinine) than non-occupationally exposed groups (23.2 μg/g creatinine). Similarly, 2-OH-Nap and 1-OH-Nap were the richest OH-PAHs, totally accounting for 82% of total OH-PAHs. In the largest e-waste recycling sites of Africa, Feldt et al. (2014) also found that levels of six OH-PAHs in the urine of e-waste exposed employees (median of 0.99 μg/g creatinine for 3-OH-Phe, 0.54 μg/g creatinine for 2-/9-OH-Phe, 0.22 μg/g creatinine for 4-OH-Phe, 0.85 μg/g creatinine for 1-OH-Nap, and 1.33 μg/g creatinine for 1-OH-Pyr), were all significantly higher than the control group as well as the urinary levels of e-waste dismantling workers in China (Lu et al., 2016).

Human hair and nail have also recently been recognized as a suitable non-invasive bioindicator of human exposure to many chemical pollutants. However, studies on PAHs in human hair and nail from e-waste dismantling areas are so far scarce. A novel method for the simultaneous detection of parent and metabolic PAH isomers was developed by our group (Lin et al., 2019), and the concentrations of PAH and OH-PAHs was firstly obtained in hair (Lin et al., 2020b) and nail samples (Ma et al., 2021) from e-waste dismantling workers as well as people near the e-waste sites. Results showed that except for Nap, levels of Σ_{15}PAHs ranged from 1.7 to 738 ng/g dw in hair samples. Phe, Fluo and Pyr were found to be the three most dominant PAH congeners (Lin et al., 2020b). The concentrations (ranging from 7.05 to 1280 ng/g, median value of 101 ng/g) and composition profiles of PAHs in match fingernail samples were comparable with those of hair sample (Ma et al., 2021). However, no significant differences in concentrations were observed between occupationally exposed workers and non-occupationally exposed people, indicating that e-waste recycling was not the only source of PAH exposure. Besides the ten regular OH-PAHs, 3-hydroxy benzo(a)pyrene (3-OH-BaP) and 6-hydroxychrysene (6-OH-Chr) were also detectable in hair and fingernail samples. The OH-PAH profile of hair samples was similar to that of paired urine samples, while 3-OH-BaP was not found in the urine samples. 2-OH-Nap and 1-OH-Nap were again obtained as the most dominant OH-PAHs in paired urine and hair samples (Lin et al., 2020b). However, 2-OH-Nap has a contribution to almost all of the monohydroxy naphthalene in fingernails, while 1-OH-Nap to $< 1\%$ (Ma et al., 2021). This might have relation to the specific metabolic characteristics and special distribution of OH-Nap in fingernails but more evidence is needed to clarify. Nevertheless, it is worth noting that OH-PAHs in nail and hair samples reflect average concentrations of long-term exposure over a period of weeks or months, whereas the levels in urine fluctuate and only stand for recent exposure (< 48 h) because of the short half-lives of PAHs in human body (Li et al., 2010). Therefore, biomonitoring with human hair and nail samples may be a more reliable way to investigate PAH metabolites.

Overall, the profiles of PAHs determined in different human biological samples from e-waste dismantling areas seem inconsistent, in line with the different biota samples used. OH-Nap was often the most important PAH metabolite, although its parent compound (Nap) was a minor PAH congener in most human tissues, possibly because of limitations in the detection accuracy of Nap due to its high volatility or the significantly different toxicokinetic processes of Nap and the predominant PAH congeners, e.g., Phe and Pyr. However, the presence of hydroxylated PAH metabolites is of more concern owing to their involvement in a series of toxicity enhancement or elimination processes of parent PAHs, which may be better reflect the risk to human health. Moreover, it should be noted that no significant discrepancy in the exposure characteristics of PAHs was observed between e-waste dismantling exposed people and reference groups, which may be affected by other exposure sources, such as cooking and traffic emissions.

2.2. PAH derivatives in various environmental matrixes and human body

Compared with the parent PAHs, there are few reports concerning the occurrences and environmental fate of derivatives of PAHs. Several derivatives related to e-waste dismantling activities have been reported from field observations and simulation studies, including Br-PAHs, Cl-PAHs, NPAHs, OPAHs, and alkylated PAHs, (Wei et al., 2012; Asamoah et al., 2019; Huang et al., 2019; Tang et al., 2020). These derivatives can be emitted from primary sources, e.g., the incomplete combustion of plastic polymers, or may originate from atmospheric secondary formation (Wei et al., 2012; Tang et al., 2020). In addition, the toxicities of some derivatives may be even higher than their precursor and could induce AhR-mediated activities (Bandowe and Meusel, 2017; Tang et al., 2020). As such, more attention need be given to the formation mechanisms and potential adverse effects of the PAH derivatives. Furthermore, the utilization of some specific derivatives may improve source identification and apportionment of PAHs (Bandowe and Meusel, 2017). We next summarize these references according to the classification of derivatives rather than the matrixes due to the limited data reported on these derivatives.

2.2.1. Alkylated PAHs

Methylated PAHs (MePAHs), such as MeNap, and methyl phenanthrene (MePhe), have been widely investigated by field observation and bench-scale pyrolysis. Chen et al. (2016) identified several MePAHs in particulate matter at urban and rural e-waste sites in South China with concentrations in the range of 0.063–3.444 ng/m^3, almost an order of magnitude lower than the levels of the parent PAHs. However, the concentrations of 1-MeNap and 2-MeNap found in dumpsite soils from Agbogbloshie were higher than that of Nap, and the contribution of these two MeNap isomers to the total $\sum_{18}$ PAHs ranged from 10% to 46% (Daso et al., 2016). The concentrations of MeNap and MePhe were also found to be higher than or comparable to those of their corresponding parent PAHs in sediments from an e-waste site (Pang et al., 2012). In addition, MeBaA and MePhe were also detected in soil and sediment samples from e-waste site of northern Vietnam (Hoa et al., 2020). Significantly higher concentrations of 2-MeNap (median, 143 ng/g lipid weight) and 1-MeNap (median, 15.8 ng/g lipid weight) have been found in human breast milk samples from the Agbogbloshie e-waste site. The concentrations of these two MeNap isomers were slightly lower than that of Nap but significantly higher than other priority PAHs (Asamoah et al., 2019). Consistent with the results observed in environmental samples, the concentrations of 2-MeNap were obtained higher than 1-MeNap in human milk. In addition, the accumulation of 2-MeNap in breast milk of mothers residing in Agbogbloshie e-waste area was much higher than the residential area (Asamoah et al., 2019). Emission of MePAHs has been demonstrated during the combustion of different plastic polymers in bench-scale reactors, with significantly lower concentrations of alkylated PAHs reported than their unsubstituted PAHs. (Wheatley et al., 1993; Panagiotou et al., 1996; Font et al., 2004; Molto et al., 2011). However, during the pyrolysis of brominated high impact polystyrene and acrylonitrile-butadiene-styrene, volatile pyrolysis products contained much higher amounts of alkylated substituted compounds (Hall et al., 2008). Nevertheless, illegal e-waste dismantling activities have caused the emission of MePAHs. As some studies have indicated a predominance of 2-OH-Nap and 1-OH-Nap in human urine from e-waste workers, the potential hydroxylated metabolites of MeNap should also be targeted in further studies.

2.2.2. NPAHs and OPAHs

Although studies have indicated that the toxic effects of several NPAHs are more pronounced than those of the parent PAHs (Bandowe and Meusel, 2017), only limited data are available concerning the occurrences of NPAHs from e-waste sites. Wei et al. (2012) reported that the concentration of NPAHs in PM$_{2.5}$ at e-waste sites in Qingyuan ranged from 302 to 406 pg/m^3 in winter and from 40.9 to 175 pg/m^3 in summer. The most abundant isomer was 3 + 2-nitrofluoranthene (3 +2 N-Fluo), followed by 9-nitroanthracene (9 N-Ant) and 7-nitrobenzo[*a*]anthracene (7 N-BaA). These three isomers comprised 65.1–88.8% of ΣNPAHs measured (Wei et al., 2012). However, during pyrolytic disposal of e-waste, 5-nitroacenaphthene (5 N-Ace) and 9-nitrophenanthrene (9 N-Phe) were found to be the predominant NPAH isomers emitted from the thermal treatment of WPCBs, while 9 N-Ant was found to be the major compound originating from the pyrolytic disposal of plastic casings (Huang et al., 2019). Different composition profiles of NPAHs may be related to the different types of dismantled plastic materials. In addition, the difference in reactivity for different NPAHs compounds might also resulted in the different composition profiles of NPAHs, since there are studies indicated that NPAHs can react with OH, NO_3 radicals, NO_2, N_2O_5 and O_3 (Bandowe and Meusel, 2017). However, further studies are needed to clarify this observation.

In addition, the occurrences of OPAHs were generally analyzed together with these NPAHs, for instance, in $PM_{2.5}$, the atmospheric concentrations of ΣOPAHs were shown to range from 519 to 930 pg/m^3 in summer and from 695 to1246 pg/m^3 in winter in an e-waste dismantling site in Qingyuan (Wei et al., 2012), which was significantly higher than NPAHs. The three main OPAH species were benzanthrone, 9,10-anthraquinone (ATQ), and 6 H-benzo[cd]pyrene-6-one, contributing 84.2–95.3% of ΣOPAHs (Wei et al., 2012). In the pyrolytic disposal treatment of e-waste, the average EFs of OPAHs were obtained as 3.37 ± 4.10 mg/g for WPCBs and 32.6 ± 18.1 mg/g for plastic castings, almost two or three orders of magnitude higher than EFs of NPAHs (85.7 ± 92.4 ng/g for WPCBs and 83.3 ± 69.7 ng/g for plastic castings) (Huang et al., 2019). 9-Fluorenone (9FLO) and ATQ were also found to be predominant isomers of OPAHs. Correlation analysis between these derivatives and their corresponding parent species indicated that the derivatives might be from incomplete combustion of the primary source (Huang et al., 2019). A significantly higher concentration of OPAHs (1.39×10^4 pg/m^3) was also found in air particles from a workshop that used an electric heating furnace to dismantle televisions, with ATQ (accounting for 83.0%) being the dominant isomer (Liu et al., 2020). However, since different isomers of NPAHs and OPAHs have been targeted in different studies, the fingerprint of PAH derivatives as a pollution characteristic of e-waste dismantling still needs to be determined in follow-up research.

2.2.3. *Halogenated PAHs (HPAHs)*

It's well known that pyrolysis treatment of e-waste is an important source of the emissions of various PAHs and their derivatives. Some chlorine-containing polymers, such as polyvinyl chloride (PVC), as well as e-waste materials containing various chlorinated and brominated flame retardants may serve as important halogen sources for the formation of HPAHs. The resulting HPAHs are more persistent in organisms and may produce greater toxicity (Tang et al., 2020). Therefore, the emission of HPAHs from e-waste dismantling sites has long been a concern. Most works in this area concerned the Cl-PAHs compared with Br-PAHs. The concentrations of Cl-PAHs in environmental media have been found to be higher than those of Br-PAHs, which might relate to the relatively shorter bond length and stronger bond of C-Cl vs. C-Br. Wang et al. (2012a) demonstrated that the concentrations of Cl-PAHs (1.0–20.1 pg/m^3) in total suspended particulate samples were higher than Br-PAHs (ND–2.3 pg/m^3). Nishimura et al. (2017) studied that the levels of total Cl-PAHs and Br-PAHs in OBS soil from Philippines, Vietnam, and Ghana ranged from 21 to 2800 ng/g dw and 5.8–520 ng/g dw, respectively. The occurrence of Br-PAHs and Cl-PAHs in surface dust of e-waste recycling workshops has also been reported (Tang et al., 2020). However, the concentrations of chlorine-substituted isomers (35.1–137 ng/g dw) were shown to be slightly lower than those of bromine-substituted isomers (60.0–399 ng/g dw). The highest level of Br-PAHs was detected in a raw material crushing workshop (399 ng/g dw), indicating physical crushing of e-waste could be another important source of Br-PAHs. Cl-PAHs were also found in samples collected from workshop floor dust (37.2–139 ng/g dw) as well as other environmental media, such as electronic shredder waste (32.3–101 ng/g dw), surface soil (up to 96.4 ng/g dw) and leaves (46.0–111 ng/g dw) at an e-waste recycling facility in China (Ma et al., 2009). Levels of Cl-PAHs in soil samples were 1–3 orders of magnitude higher than those reported in sediment samples (up to 2.89 ng/g dw) from Qingyuan in southern China (Gonzales, 2011), soil (up to 3.96 ng/g dw) and river sediment (up to 3.93 ng/g dw) from Vietnam (Wang et al., 2016). Overall, the concentrations of these HPAHs were two or three orders of magnitude lower than their parent PAHs.

Different from the routine analysis of 16 parent PAHs, the congeners of HPAHs targeted by different reports vary significantly. Thus, the compositions of substituted HPAH isomers are more complicated than those of their parent precursors. The sources and compositions of several HPAHs isomer are summarized in Table 3. In general, the production of HPAHs is consistent with the active sites on the benzene ring of PAHs. For example, 1-chloropyrene (1-Cl-Pyr) is found to be one of the richest Cl-PAHs present in various environmental matrices. Gonzales (2011) showed that the HPAH profile in particulate matter at a Qingyuan e-waste site primarily comprised 1-Cl-Pyr and 6-chlorobenzo(a)pyrene (6-Cl-BaP), whereas a house dust profile primarily comprised 3-chlorofluoranthene (3-Cl-Fluo) and 1-Cl-Pyr. Tue et al. (2017) also found that 1-Cl-Pyr was the richest species among the identified Cl-PAHs and Br-Phe/Ant was found to be the most abundant compound in soil samples from e-waste burning areas. Wang et al. (2016) found that the dominant compounds were 1-Cl-Pyr and 9-chlorophenanthrene (9-Cl-Phe) in river sediments and surface soils from e-waste recycling areas. Tang et al. (2020) showed that bromine-substituted isomers with four rings (Br-BaA and Br-Pyr) were major compounds in indoor dust in e-waste workshops, whereas chlorine-substituted isomers with small rings (Cl_2-Phe and Cl-Phe/-Ant) were the predominant Cl-PAH compounds. However, information on Br-PAH profiles in the environment is still limited. Nishimura et al. (2017) reported that Cl-PAHs (Cl-Phe, Cl-Ant, and Cl-Fluo) with three rings were the predominant isomers and 9-bromophenanthrene (9-Br-Phe) and 9,10-dibromoanthracene (9,10-Br_2-Ant) were the predominant Br-PAH congeners in OBS soil. Liu

Table 3

Compositions of halogenated PAHs (HPAHs) from different sources.

Sources	Composition Profiles	Refs
Cl-PAHs and Br-PAHs from e-waste site		
Electronic shredder waste	1-Cl-Pyr (24%) ≈ 6-Cl-BaP (23%) ≈ 8-Cl-Flu (22%) > 7-Cl-BaA (17%) > 3,9,10-Cl_3-Phe	(Ma et al., 2009)
E-waste recycling facility floor-dust	6-Cl-BaP (43%) > 7-Cl-BaA (19%) > 1-Cl-Pyr (16%) > 8-Cl-Flu (10%) > 3,9,10-Cl_3-Phe	(Ma et al., 2009)
E-waste recycling facility soil	6-Cl-BaP (49%) > 1-Cl-Pyr (19%) > 3,9,10-Cl_3-Phe (13%) > 7-Cl-BaA (8%) ≈ 8-Cl-Flu	(Ma et al., 2009)
E-waste leaf samples	1-Cl-Pyr (32%) > 9-Cl-Phe (20%) > 3-Cl-Flu (10%) ≈ 7-Cl-BaA (10%)	(Ma et al., 2009)
E-waste particulate matter	6-Cl-BaP > 9-Cl-Ant > 2-Cl-Ant > 1-Cl-Pyr > 8-Cl-Fluo; 6-Br-BaP > 1-Br-Pyr	(Chen et al., 2016)
Agbogbloshie OBS soil	Cl-Phe/Ant > Cl-Flu/Pyr > Cl_2-Phe/Ant > Cl_2-Flu/Pyr > Cl_3-Phe/Ant; Br_2-Phe/Ant > Br-Phe/Ant > 1-Br-Pyr	(Tue et al., 2017)
Vietnam OBS soil	9-Cl-Phe > 3,9-Cl_2-Phe ≈ 9,10-Cl_2-Ant > 1-Cl-Pyr > 3-Cl-Flu; 1,5-Br_2-Ant > 9,10-Br_2-Ant > 2,6-Br_2-Ant > 1-Br-Ant > 9-Br-Phe	(Wang et al., 2016)
OBS soil	9,10-Cl_2-Ant > 1-Cl-Pyr > 7-Cl-BaA > 9-Cl-Ant > 9-Cl-Phe > 3,9-Cl_2-Phe > 6-Cl-BaP; 1,5-Br_2-Ant > 9,10-Br_2-Ant > 9-Br-Phe > 1-Br-Ant > 1-Br-Pyr > 7-Br-BaA	(Nishimura et al., 2017)
E-waste dust	2-Cl-Ant > 2-Cl-Phe > 9-Cl-Ant > 9,10-Cl_2-Phe > 1,5-Cl_2-Ant > 1-Cl-Pyr; 1-Br-Pyr > 1,6-Br_2-Pyr > 7-Br-BaA > 4-Br-Pyr > 1,2-Br_2-Acy > 9-Br-Phe	(Tang et al., 2020)
E-waste air particles	9,10-Cl_2-Phe > 1,5-Cl_2-Ant > 1-Cl-Pyr > 7-Cl-BaA; 1,8-Br_2-Ant > 9,10-Br_2-Phe > 3-Br-Flu	(Liu et al., 2020)
Cl-PAHs and Br-PAHs from other sources		
SRM 2585	9-Cl-Phe > 1-Cl-Pyr > 9-Cl-Ant > 1,5-Cl_2-Ant > 9,10-Cl_2-Phe > 2-Cl-Phe; 5-Br-Ace > 9-Br-Phe	(Tang et al., 2020)
Secondary copper smelting gas	9-Cl-Phe/2-Cl-Phe > 1,5-Cl_2-Ant/9,10-Cl_2-Ant > 3-Cl-Flu > 1-Cl-Pyr; 9-Br-Phe (38%) > 3-Br-Flu	(Jin et al., 2017)
Municipal waste incinerator gas	6-Cl-BaP (31.2%) > 1-Cl-Pyr (12.8%) > 3-Cl-Flu (8.4%) ≈ 7-Cl-BaA (8.1%); 1-Br-Pyr > 6-Br-BaP > 7-Br-BaA	(Horii et al., 2008)
Chlorine-chemical industrial soil	6-Cl-BaP (84%) > 1-Cl-Pyr (7%) > 7-Cl-BaA (4%) > 3,9,10-Cl_3-Phe (2%)	(Ma et al., 2009)

et al. (2020) found that air particle compositions of Br-PAHs varied significantly among different e-waste workshops, which might relate to differences in the formation and emission of Br-PAHs during e-waste dismantling activities. 1-Cl-Pyr was obtained as the most widely detected isomer. However, these results need to be treated with caution, as Cl-PAH sources other than e-waste may also be one of the reasons. In contrast, the results suggest that Br-PAHs mainly originates from e-waste recycling processes, and therefore it may be useful to include them in future monitoring programs.

3. Emission characteristics and source identification of PAHs and their derivatives

The emission of PAHs associated with the products of incomplete combustion from different types of plastics have attracted increasing concern since municipal solid waste incinerators were widely installed from the 1990 s (Wheatley et al., 1993; Li et al., 2001). The early studies mainly focused on simulating PAH-like toxic organic compounds produced by the incomplete combustion of various polymer plastics during high temperature incineration in bench-scale furnaces. Emission factors and pollution characteristics of PAHs and their derivatives from the combustion or pyrolysis processes of different synthetic polymers are summarized in Table 4. Different plastic materials used in daily life, including PVC, polyethylene (PE), low density poly(ethylene) (LDPE), high density poly(ethylene) (HDPE), poly(propylene) (PP), poly(styrene) (PS), and poly(ethylene terephthalate) (PET), were widely investigated (Wheatley et al., 1993; Panagiotou et al., 1996; Li et al., 2001; He et al., 2015). Different types of printed wiring board and plastic casing from e-waste materials or their powder mixtures were also recently employed to investigate emissions of hazardous PAHs from legal and illegal e-waste treatment (Gu et al., 2010; Font et al., 2011; Cai et al., 2018). A series of complex combustion or pyrolysis processes occurs during thermal decomposition of these synthetic polymers, involving chemical reaction, melting, evaporation, thermal degradation, heterogeneous oxidation, gas diffusion, and mass transfer (Valavanidis et al., 2008). It has been confirmed that the pyrolysis and combustion processing of e-waste is the main reason for the massive emission of various VOCs and PAHs (Wong et al., 2007; Zhang et al., 2011; An et al., 2014; He et al., 2015; Chen et al., 2019).

3.1. The effect of temperature on PAHs emissions

Temperature has been shown to be the most critical factor affecting PAH emissions from laboratory-scale pyrolysis studies. Typical operation temperatures in the literature range from 320 °C to 1200 °C (Table 3). According to different temperatures, these simulation studies can be classified into three categories: i) simulated plastic incineration at temperatures higher than or equal to 800 °C, which is the temperature reached in the post-combustion zone in combustion treatment plants (Soler et al., 2017), ii) simulated pyrolysis processes at temperatures of 400–800 °C, representing the open-air burning of e-waste, and iii) simulated pyrolysis processes at temperatures lower than 400 °C, representing the broad baking of e-waste.

In these simulated incineration and open burning studies, as the heating temperature and residence time of the reactor rose, the amount of PAHs usually decreased (Wheatley et al., 1993; Panagiotou et al., 1996). In addition, large-ring PAHs may easily decompose to form LMW PAHs or other lighter volatile compounds as the pyrolysis temperature is increased (Hall et al., 2008; He et al., 2015). Meanwhile, gas production (stack flue gas) increases and oil (bottom ash) production decreases (Herrera et al., 2002; Font et al., 2004). Herrera et al. (2002) investigated the formation of PAHs from thermal degradation of thermoplastic polyurethane elastomers and showed that the generation of PAHs was dependent on the heating temperature: higher concentrations of PAHs were observed at 950 °C than that at 800 °C. Furthermore, the chemical structure of polyurethanes polymers with different long chain diols

Table 4
Emission factors and pollution characteristics of PAHs and their derivatives from the combustion or pyrolysis processes of different synthetic polymers.

Plastic[a]	Temperature	Emission Factor	Characteristics of PAHs	Refs
PVC	450–850 °C[b] combustion	flue gas 320.5 mg/kg & bottom ash 195.4 mg/kg;	Nap, Acy, Ace, and Flu dominated; HMW PAHs constituted a larger percentage in the bottom ash from incineration of PVC than PP and HDPE.	(Li et al., 2001)
HDPE		flue gas 462.3 mg/kg & bottom ash 71.4 mg/kg;		
PP		flue gas 315.3 mg/kg & bottom ash 45.2 mg/kg;		
TPU	800 °C[c] 950 °C[c]	≈ 20 mg/kg ≈ 40 mg/kg	Nap > Phe > Acy > Pyr	(Herrera et al., 2002)
PS	600–750 °C combustion	particulate soot 912 mg/kg & solid ash 509 mg/kg	Nap > B[k]F > Flu	(Valavanidis et al., 2008)
PVC		particulate soot 1409 mg/kg & solid ash 1088 mg/kg	Nap > Bap > B[b]F	
LDPE		particulate soot 747 mg/kg & solid ash 402 mg/kg	Nap > B[k]F > Flu	
HDPE		particulate soot 765 mg/kg & solid ash 431 mg/kg	Bap > B[k]F > Nap	
PP		particulate soot 625 mg/kg & solid ash 269 mg/kg	Nap > B[k]F > Ace	
PET		particulate soot 363 mg/kg & solid ash 319 mg/kg	Nap > Ace > Flu	
PCA	450 °C combustion	121 mg/kg	Nap predominated (> 80%)	(Chien et al., 2010)
PCL	400 °C combustion	10.55 mg/kg	Nap (27.9%) > Phe (26.7%) > D[a,h]A (7.7%)	(Chien and Yang, 2013)
FPUF	850 °C pyrolysis 550 °C pyrolysis 850 °C combustion 550 °C combustion	6871 mg/kg 191 mg/kg 18 mg/kg 44 mg/kg	Nap > Acy > Phe > Pyr > Flu	(Garrido et al., 2016)
Mobile phone case	500 °C pyrolysis 500 °C combustion	244 mg/kg 224 mg/kg	Nap > Ant > Flu > Phe Nap > Ant > Pyr	(Font et al., 2011)
Printed circuit board	850 °C pyrolysis	8630 mg/kg	Nap (64%) > Phe (15%) > Acy (4%)	(Molto et al., 2011)
	850 °C combustion	5288 mg/kg	Nap (51%) > Phe (16%) > Flu (8%)	
Mobile phone case	850 °C pyrolysis	44,098 mg/kg	Nap (38%) > Acy (24%) > Flu (13%) > Phe (12%)	
	850 °C combustion	40,717 mg/kg	Nap (31%) > Acy (24%) > Flu (16%) > Phe (12%)	

(continued on next page)

Table 4 (*continued*)

Plastic[a]	Temperature	Emission Factor	Characteristics of PAHs	Refs
Electronic waste	850 °C pyrolysis	62,403 mg/kg	Nap (37%) > Acy (25%) > Phe (16%)	
	850 °C combustion	55,337 mg/kg	Acy (32%) > Nap (29%) > Phe (15%)	
Printed circuit board	850 °C pyrolysis	32,627 mg/kg	Nap (56%) > Acy (22%) > Phe (11%)	(Soler et al., 2017)
Printed circuit board	320 °C combustion	2.77 mg/kg	Nap (42%) > Phe (22%) > Flu (15%)	(Cai et al., 2018)
Plastic casings		23.65 mg/kg	Nap (53%) > Phe (18%) > Flu (6%)	
Printed circuit board	320–600 °C combustion	2.28–12 mg/kg	Nap > Phe ≈ Flu	(Cai et al., 2019)
Printed circuit board	320 °C combustion	OPAHs: 3.37 mg/kg NPAHs: 0.086 mg/kg	9FLO (69%) > ATQ (27%) 5 N-ACE (64%) > 9 N-Phe (12%)	(Huang et al., 2019)
Plastic casings		OPAHs: 32.6 mg/kg NPAHs: 0.083 mg/kg	9FLO (54%) > ATQ (24%) 9 N-ANT (30%)	

[a] polyvinyl chloride, HDPE = high density polyethylene, PP = polypropylene, TPU = thermoplastic polyurethane, PS = polystyrene, LDPE = low density polyethylene, PET = polyethylene terephthalate, PCA = polylactic acid, PCL = polycaprolactone, FPUF = flexible polyurethane foam.

[b] Temperatures for the first (450 °C) and second (850 °C) combustion chambers.

[c] Similar results were observed under the pyrolysis and combustion treatment.

showed similar PAH-concentrations in the evolved gas. In addition, different carrier gases, such as nitrogen and synthetic air, did not seem to have a much impact on the formation of PAHs. However, Hall et al. (2008) reported that as reaction temperature increased from 410 °C to 520 °C, multiple-ring aromatics from the pyrolysis of flame-retarded plastics, e.g., acrylonitrile butadiene styrene (ABS) and high-impact polystyrene (HIPS), were more effectively converted into lighter pyrolysis products. In addition, during combustion of flexible polyurethane foam (FPUF), pyrolysis at 850 °C resulted in substantially higher emissions of PAHs (6871 ± 76 mg/kg sample) than those at 550 °C (191 ± 32 mg/kg sample). Conversely, during the incineration of FPUF, the yields of PAHs at 550 °C (44 ± 5 mg/kg sample) were relatively higher than those at 850 °C (18 ± 2 mg/kg sample) (Garrido et al., 2016). Excess amounts of oxygen supplied during the combustion of waste plastics was found to decrease both PAH concentrations in bottom ash and the EFs of total PAHs in the stack flue gas (Li et al., 2001). On the other hand, during broad baking studies, high temperature increased the EFs of gaseous PAHs but also had detectable effects on the EFs of particulate PAHs. Nevertheless, PAHs released from baking WPCBs under relatively low heat conditions were substantially lower than those from incineration and open burning of e-waste. The newly emitted PAHs mainly existed in gaseous phase at ≥ 520 °C, whereas particulate phase PAHs prevailed at low temperatures (Cai et al., 2019).

3.2. *The effect of different plastics on PAHs emissions*

The EFs for the pyrolysis of different types of plastics or e-waste component categories have been shown to vary substantially. In general, the controlled combustion of PVC produces the highest amounts of PAHs compared with other types of plastic (Li et al., 2001; Valavanidis et al., 2008), whereas the amounts of PAHs emitted from the combustion of polycaprolactone (PCL) and thermoplastic polyurethane (TPU) were lower than other plastics (Herrera et al., 2002; Chien and Yang, 2013). Similar distribution profiles of emitted individual PAHs were found after incinerating PP and HDPE wastes (Li et al., 2001), which might be related to the similar physical and chemical properties of these two plastic polymers.

From studies of burning different e-waste component categories, EFs for the combustion of plastic casings were shown to be higher than those of mobile phone cases, and both were much higher compared with those of WPCBs (Molto et al., 2011; Cai et al., 2018) (Table 3). In addition, several studies have investigated the mass of PAHs discharged as evolved gaseous products or ash residue and shown that most PAHs are emitted mainly in the stack flue gas or particulate soot (Li et al., 2001; Valavanidis et al., 2008). LMW PAHs, including Nap, Phe, Acy, Ace and Flu, are the predominant isomers found in combustion exhausts. However, the concentrations of HMW PAHs in the residual solid ash were higher than those in particulate soot emitted from burning PVC plastic (Valavanidis et al., 2008). Relatively higher concentrations of HMW PAHs were also detected in the residual solid ash from combusting PVC than those from other types of plastics (Panagiotou et al., 1996; Valavanidis et al., 2008). In addition, the pyrolysis or combustion of various e-wastes emitted almost exclusively 2–3 rings PAHs, with contributions of more than 80% (Font et al., 2011; Molto et al., 2011; Soler et al., 2017; Cai et al., 2018), whereas HMW PAH isomers with fused rings, such as BaP, B[*k*]F and B[*b*]F, were often reported during the incineration of various polymer plastics. Specifically, as PVC plastic wastes were combusted, HMW PAHs accounted for a great proportion in the bottom ash than those from HDPE and PP plastics (Li et al., 2001). Substituted PAHs, chlorinated aromatics and oxygenated compounds have also been detected in the combustion products of PVC (Panagiotou et al., 1996). One study reported the greater mutagenicity of extracts of particulates from the combustion of PVC compared with PET, PS, and PE (Lee et al., 1995). Therefore, more research is needed to investigate the toxic aromatic compounds released from combustion of PVC instead of other plastics.

3.3. *The formation mechanism of PAHs and their derivatives*

It has been concluded that the types of aromatic hydrocarbon emissions rely on the types of polymer plastic burned, whereas their relative amounts are effected by the combustion parameters including residence time, presence of oxygen, and temperature (Panagiotou et al., 1996). However, the pyrolysis parameters of plastics vary considerably and are difficult to replicate among the different studies. Nevertheless, several studies have proposed a possible formation mechanism of PAHs and their derivatives during the incineration and pyrolysis of plastic fragments of e-waste materials, well known as the precursor theory (Herrera et al., 2002; Garrido et al., 2016). This theory has been used to interpret the congener distributions of PAHs after thermal treatment, and it assumes that the first aromatic ring (benzene) is formed and then begins to grow to form HMW PAHs. Therefore, naphthalene concentration is higher than that of phenanthrene, whose level is higher than that of pyrene (Herrera et al., 2002; Garrido et al., 2016).

On the other hand, several studies have also investigated the formation of TPB during the incineration of plastics, such as burning polyethylene plastics (Simoneit et al., 2005; Tomsej et al., 2018), as well as thermal pyrolysis of HIPS (at 410 °C or 520 °C) and ABS (only at 410 °C) (Hall et al., 2008). TPB has also been detected in particles generated from burning hard plastic blocks but not from burning wires/cables (Gu et al., 2010). The symmetric 1,3,5-isomer was found to be the dominant congener, along with minor products of 1,2,4-triphenylbenzene (Simoneit et al., 2005; Gu et al., 2010; Tomsej et al., 2018). The formation mechanism of these compounds varied from the conventional mechanism of PAHs and might have involved dehydration/cyclization of the polyoxyethylene moieties of some antioxidants, as proposed by Simoneit et al. (2005). Apart from TPB and its isomers, five abundant unknown compounds were also tentatively detected by mass spectrometry at m/z 306, corresponding to the base peak of TPB. Correlation analysis showed a strong correlation of TPB with these five unknown compounds, indicating that these contaminants may have the same source as TPB (Tomsej et al., 2018). In addition, the derivatives such as OPAHs and NPAHs were also emitted from the pyrolytic disposal of WPCBs (Huang et al., 2019). Correlation analysis further indicated that emissions of OPAHs could be used as potential tracers of e-waste dismantling by considering similar concentrations levels against their parent PAHs (Huang et al., 2019). However, more evidence from field studies is needed to verify the specific application of these tracers for e-waste pollution.

3.4. Source apportionment of PAHs

The specific diagnostic ratios of paired PAH isomers, including Ant/(Ant + Phe), B[*a*]A/(B[*a*]A + Chr), Fluo/(Fluo + Pyr), IcdP/(IcdP + B[ghi]P), B[*b*]F/(B[*b*]F + B[*k*]F), and B[*a*]P/(B[*a*]P + B[ghi]P), are frequently used in identifying sources of PAHs. The use of these ratios is based on the hypothesis that PAH isomers present the same physicochemical properties and will consequently be transformed and degraded at the same rate, preserving the ratio values of the emission sources. The ratios of these six paired isomers reported in atmospheric particles, soil, and sediment samples from e-waste dismantling areas and those from the pyrolytic treatment of various plastics and e-waste are summarized in Table S1. As seen, the diagnostic ratios of PAHs from e-waste dismantling activities lie within the same range or overlap with the threshold values of other emission sources. As such, there is a risk of misdiagnosis when using these ratios, and new threshold values of isomer ratios that may distinguish between e-waste pyrolysis and other sources need to be proposed. Household coal is frequently used as stove fuel for baking WPCBs (also known as board baking), which may lead to a mixed emission of PAHs originating from plastic and coal combustion during e-waste recycling, further increasing the difficulty of source apportionment of PAH pollutants (Zhang et al., 2011). Some works have demonstrated mixed sources of PAHs from the e-waste areas (Chen et al., 2019; Xu et al., 2016). Therefore, a diagnostic ratio that represents both the sources from plastic combustion and coal-fired emission may be a better indicator of e-waste pollution. As summarized in Table S1, B[*b*]F/(B[*b*]F + B[*k*]F) > 0.5 and IcdP/(IcdP + B[ghi]P) > 0.4 may be used as preliminary diagnostic ratios to distinguish e-waste sources from other emissions. In addition, B[*a*]A/(B[*a*]A+B[ghi]P) and IcdP/(IcdP + B[ghi]P) have been proposed as preliminary diagnostic ratios to discriminate other emission sources from pyrolysis of e-wastes (Cai et al., 2018). Furthermore, it is noteworthy that a large number of PAHs and/or their derivatives have been reported besides the 16 priority PAHs during primitive e-waste dismantling activities, which could also be used as specific indicators or diagnostics for source appointment. However, further research is needed.

4. Health consequences of exposure to PAHs

Inhalation is one of the important exposure pathways to environmental pollutants for e-waste dismantlers. Luo et al. (2015) studied the effect of particle-bound PAHs on human health in a typical e-waste dismantling area. Their results suggested that PAHs deposited in different respiratory tract regions depended on particulate size, and accumulation mode particles (Dp:0.10–1.8 μm) were mainly responsible for the carcinogenic toxicity of PAHs in the lung, which is the greatest contribution to the cancer risk. Wang et al. (2012a) reported that residents near the e-waste sites have a more serious inhalation cancer risk because of PAH exposure, especially MW 302 PAHs. However, only a few studies have been conducted on the molecular and cellular level health effects caused by carcinogenic or HMW PAHs emitted from e-waste recycling areas. Some works have demonstrated a relationship between general PAH exposure and oxidative stress in people in e-waste recycling areas. Alabi et al. (2012) measured DNA damage in human peripheral blood lymphocytes following treatment with leachates of soil and plants from e-waste processing sites. Their results suggested that PAHs were responsible for DNA damage. Positive association of human urinary concentrations of OH-PAHs with levels of two biomarkers of oxidative stress (malondialdehyde (MDA) and 8-hydroxy-2′-deoxyguanosine (8-OHdG)) also indicated that the exposure to PAHs in an e-waste recycling area might cause DNA damage (Lu et al., 2016). However, large amounts of dioxin-like chemicals other pollutants emitted during e-waste dismantling activities may also cause similar DNA damage (Wen et al., 2008), complicating the interpretation of results.

The effects of PAH exposure on the health of susceptible populations, such as pregnant women and children, at e-waste dismantling sites has received wide concern. The insulin-like growth factor (IGF)-binding protein (IGFBP) is an important system for assessing fetal growth. Xu et al. (2013) found that levels of $\Sigma_{5\text{ring}}$-PAHs in umbilical cord blood at an e-waste dismantling site were associated with elevated placental IGF-1, while levels of four-ring and total PAHs in umbilical cord blood correlated with IGFBP-3 expression, which has an indirect effect on fetal growth and development. However, Zeng et al. (2020) suggested that a decreased plasma IGF-1 level mediated 23.9% of the whole effect associated with $PM_{2.5}$-bound Σ_{16} PAHs exposure on child height. In other words, an elevation of 1 ng/kg of $PM_{2.5}$-bound Σ_{16} PAHs intake per day was associated with a 0.022 cm decrease of child height. Huo et al. (2019) also suggested that high exposure to PAHs during the pregnancy in e-waste areas could pose a threat to neonatal development based on association of OH-PAH concentrations in maternal urine with parameters of birth outcomes. Xu et al. (2015) found that the effect of PAH exposure in e-waste areas on fetal growth was more significant in boys.

The effects of PAH exposure on inflammation and cardiovascular disease risk in preschool children were also investigated in e-waste recycling areas. Cheng et al. (2020) found that increased PAH exposure was associated with higher aryl hydrocarbon receptor (AhR) and NLR family pyrin domain containing 3 (NLRP3) expression and eventually mediated the levels of serum cytokines to increase the risk of inflammatory infection. Dai et al. (2019) found significant negative correlations between the levels of mean platelet volume (MPV), platelet distribution width (PDW), platelet-large cell ratio (P-LCR) and ratio of mean platelet volume to platelet count (MPVP) with OH-PAHs, which were associated with low-grade inflammation in children in e-waste dismantling areas. Furthermore, Zheng et al. (2019) also provided the evidence of a relationship between Pb/PAH co-exposure and cardiovascular endothelial inflammation in e-waste exposed preschoolers. They found that children with higher co-exposure concentrations of PAHs and Pb showed aggravated damage of the vascular endothelium by the modulation of inflammatory processes, which suggested a potential future cardiovascular disease risk. However, further studies are still needed to determine the damage effect as well as damage mechanism of the co-exposure of heavy metals with PAHs on human health, particularly as heavy metals are the most important inorganic pollutants emitted from e-waste recycling process.

5. Conclusions and perspective

This review mainly summarizes the environmental distribution, fate, emission characteristics and formation mechanisms of PAHs and their derivatives in air, soil, biota as well as human biological samples from e-waste recycling sites. Data show that atmospheric PAH levels in typical e-waste sites, e.g., Guiyu from Sout2hern China, have sharply declined over the last few decades. Similarly, the concentrations of PAHs in soil from old e-waste sites in China have also decreased, which are generally lower than those from newly emerged sites in India, Pakistan and Ghana. The emission of PAHs is associated with the incomplete combustion of different e-waste materials. Compared with other types of plastic, e-waste containing PVC produces the highest amounts of PAHs during controlled combustion process. However, it is difficult to identify the e-waste characteristic source of PAHs based solely on the profile of the 16 U.S. EPA priority PAHs and the specific diagnostic ratios of the paired PAH isomers due to wide usage of honeycomb coals for informal e-waste dismantling and complication of other exposure sources, such as cooking and traffic emissions. Fortunately, MW 302 PAHs, TPB as well as some derivatives of PAHs, such as Cl-PAHs, Br-PAHs and OPAHs, have been found to be potential specific tracers to improve source apportionment and allocation of PAHs from the e-waste recycling. Since it may be difficult to unambiguously identify the characteristics and toxicities of PAHs and their derivatives from e-waste sites, more following research is still needed to clarify the profiles of PAH pollutants and potential harmful health consequences of e-waste dismantling activities in the near future.

Firstly, HMW PAHs, such as MW 302 PAHs and TPB, should be listed as e-waste priority PAH isomers. Much attention should be paid on PAH derivatives, particularly alkylated PAHs, Br-PAHs and OPAHs in various environmental matrices at e-waste sites. More research is needed for clarification of their pollution profiles and formation mechanism for further application in more reliable source appointment.

Secondly, to evaluate the comprehensive ecology and human health risks of PAH exposure from e-waste dismantling areas, the distribution characteristics of MW 302 PAHs, TPB, some derivatives of PAHs and their metabolites in biota and human tissues should be explored more extensively, especially the distribution of PAHs in biota are found to be species- and tissues-dependent. Special attention should be paid to their transformation products or metabolites in order to identify potential biomarkers for tracing the effect of these emitted pollutants on human being resulted from e-waste pollution.

Finally, besides PAHs, humans may also be co-exposed to other toxic pollutants, such as heavy metals or VOCs emitted from e-waste recycling processes. In the presence of other compounds, e.g., heavy metals, synergistic, additive or antagonistic effects may occur in the biotransformation mechanisms and toxicity effects of PAHs in the human body. Therefore, toxicity mechanisms of combined pollutants in the human body should also be studied in detail to answer the risk of PAHs exposure from e-waste dismantling activities.

Novelty statement

Inappropriate e-waste dismantling practices have caused the emission of toxic substances and thus become an urgent global issue due to the potential adverse consequences of human exposure. Among them, PAHs are of great concern due to their mutagenic and carcinogenic potencies to humans. Thus, this review aims to provide an overview in the current knowledge on the occurrences of PAHs in various environmental matrices from e-waste dismantling areas and their emission characteristics, source analysis, formation mechanism, human body burdens and potential health consequences. This study would provide a scientific basis for implementing pollution regulations and emission controls.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.jhazmat.2021.127683.

References

Alabi, O.A., Bakare, A.A., Xu, X., Li, B., Zhang, Y., Huo, X., 2012. Comparative evaluation of environmental contamination and DNA damage induced by electronic-waste in Nigeria and China. Sci. Total Environ. 423, 62–72.

An, T., Huang, Y., Li, G., He, Z., Chen, J., Zhang, C., 2014. Pollution profiles and health risk assessment of VOCs emitted during e-waste dismantling processes associated with different dismantling methods. Environ. Int. 73, 186–194.

Asamoah, A., Fini, M.N., Essumang, D.K., Muff, J., Sogaard, E.G., 2019. PAHs contamination levels in the breast milk of Ghanaian women from an e-waste recycling site and a residential area. Sci. Total Environ. 666, 347–354.

Awasthi, A.K., Zeng, X.L., Li, J.H., 2016. Environmental pollution of electronic waste recycling in India: a critical review. Environ. Pollut. 211, 259–270.

Bandowe, B.A.M., Meusel, H., 2017. Nitrated polycyclic aromatic hydrocarbons (nitro-PAHs) in the environment – a review. Sci. Total Environ. 581–582, 237–257.

Bi, X., Sheng, G., Peng, P., Zhang, Z., Fu, J., 2002. Extractable organic matter in PM10 from LiWan district of Guangzhou City, PR China. Sci. Total Environ. 300, 213–228.

Cai, C.Y., Yu, S.Y., Li, X.Y., Liu, Y., Tao, S., Liu, W.X., 2018. Emission characteristics of polycyclic aromatic hydrocarbons from pyrolytic processing during dismantling of electronic wastes. J. Hazard. Mater. 351, 270–276.

Cai, C.Y., Chen, L.Y., Huang, H.J., Liu, Y., Yu, S.Y., Liu, Y., Tao, S., Liu, W.X., 2019. Effects of temperature on the emission of particulate matter, polycyclic aromatic hydrocarbons, and polybrominated diphenyl ethers from the thermal treatment of printed wiring boards. J. Hazard. Mater. 380, 120849.

Chakraborty, P., Sampath, S., Mukhopadhyay, M., Selvaraj, S., Bharat, G.K., 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. Environ. Pollut. 248, 1036–1045.

Chen, A., Dietrich, K.N., Huo, X., Ho, S.M., 2011. Developmental neurotoxicants in e-waste: An emerging health concern. Environ. Health Perspect. 119, 431–438.

Chen, H.J., Ma, S.T., Yu, Y.X., Liu, R.R., Li, G.Y., Huang, H.B., An, T.C., 2019. Seasonal profiles of atmospheric PAHs in an e-waste dismantling area and their associated health risk considering bioaccessible PAHs in the human lung. Sci. Total Environ. 683, 371–379.

Chen, L., Yu, C.N., Shen, C.F., Zhang, C.K., Liu, L., Shen, K.L., Tang, X.J., Chen, Y.X., 2010. Study on adverse impact of e-waste disassembly on surface sediment in East China by chemical analysis and bioassays. J. Soil. Sediment. 10, 359–367.

Chen, S.J., Wang, J., Wang, T., Wang, T., Mai, B.X., Simonich, S.L.M., 2016. Seasonal variations and source apportionment of complex polycyclic aromatic hydrocarbon mixtures in particulate matter in an electronic waste and urban area in South China. Sci. Total Environ. 573, 115–122.

Cheng, Z., Huo, X., Dai, Y., Lu, X., Hylkema, M.N., Xu, X., 2020. Elevated expression of AhR and NLRP3 link polycyclic aromatic hydrocarbon exposure to cytokine storm in preschool children. Environ. Int. 139, 105720.

Chien, Y.C., Yang, S.H., 2013. Investigation of the combustion kinetics and polycyclic aromatic hydrocarbon emissions from polycaprolactone combustion. Environ. Technol. 34, 149–155.

Chien, Y.C., Liang, C.J., Liu, S.H., Yang, S.H., 2010. Combustion kinetics and emission characteristics of polycyclic aromatic hydrocarbons from polylactic acid combustion. J. Air Waste Manag. 60, 849–855.

Dai, Y., Huo, X., Cheng, Z., Wang, Q., Zhang, Y., Xu, X., 2019. Alterations in platelet indices link polycyclic aromatic hydrocarbons toxicity to low-grade inflammation in preschool children. Environ. Int. 131, 105043.

Daso, A.P., Akortia, E., Okonkwo, J.O., 2016. Concentration profiles, source apportionment and risk assessment of polycyclic aromatic hydrocarbons (PAHs) in dumpsite soils from Agbogbloshie e-waste dismantling site, Accra, Ghana. Environ. Sci. Pollut. Res. 23, 10883–10894.

Deng, W.J., Louie, P.K.K., Liu, W.K., Bi, X.H., Fu, J.M., Wong, M.H., 2006. Atmospheric levels and cytotoxicity of PAHs and heavy metals in TSP and PM2.5 at an electronic waste recycling site in southeast China. Atmos. Environ. 40, 6945–6955.

Feldt, T., Fobil, J.N., Wittsiepe, J., Wilhelm, M., Till, H., Zoufaly, A., Burchard, G., Goeen, T., 2014. High levels of PAH-metabolites in urine of e-waste recycling workers from Agbogbloshie, Ghana. Sci. Total Environ. 466, 369–376.

Font, R., Aracil, I., Fullana, A., Conesa, J.A., 2004. Semivolatile and volatile compounds in combustion of polyethylene. Chemosphere 57, 615–627.

Font, R., Molto, J., Egea, S., Conesa, J.A., 2011. Thermogravimetric kinetic analysis and pollutant evolution during the pyrolysis and combustion of mobile phone case. Chemosphere 85, 516–524.

Gao, Y.Y., Wang, Y.Y., Zhou, Q.X., 2015. Distribution and temporal variation of PCBs and PAHs in soils and sediments from an e-waste dismantling site in China. Environ. Earth Sci. 74, 2925–2935.

Garrido, M.A., Font, R., Conesa, J.A., 2016. Pollutant emissions during the pyrolysis and combustion of flexible polyurethane foam. Waste Manag. 52, 138–146.

Gonzales, L.R., 2011. Trace Analysis of Halogenated Polycyclic Aromatic Hydrocarbons from an Electronic Waste Recycling Area and Guangzhou. China. E-waste. Oregon State University (Master's Thesis).

Grant, K., Goldizen, F.C., Sly, P.D., Brune, M.N., Neira, M., van den Berg, M., Norman, R. E., 2013. Health consequences of exposure to e-waste: a systematic review. Lancet Glob. Health 1, E350–E361.

Gu, W.H., Bai, J.F., Yuan, W.Y., Ma, E., Zhang, C.L., Wang, J.W., 2019. Pollution analysis of soil polycyclic aromatic hydrocarbons from informal electronic waste dismantling areas in Xinqiao, China. Waste Manag. Res. 37, 394–401.

Gu, Z., Feng, J., Han, W., Wu, M., Fu, J., Sheng, G., 2010. Characteristics of organic matter in PM2.5 from an e-waste dismantling area in Taizhou, China. Chemosphere 80, 800–806.

Guo, Y., Huo, X., Wu, K., Liu, J., Zhang, Y., Xu, X., 2012. Carcinogenic polycyclic aromatic hydrocarbons in umbilical cord blood of human neonates from Guiyu, China. Sci. Total Environ. 427, 35–40.

Hall, W.J., Miskolczi, N., Onwudili, J., Williams, P.T., 2008. Thermal processing of toxic flame-retarded polymers using a waste fluidized catalytic cracker (FCC) catalyst. Energy. Fuel 22, 1691–1697.

He, M.J., Yang, S.Y., Zhao, J., Collins, C., Xu, J.M., Liu, X.M., 2019. Reduction in the exposure risk of farmer from e-waste recycling site following environmental policy adjustment: a regional scale view of PAHs in paddy fields. Environ. Int. 133, 105136.

He, Z., Li, G., Chen, J., Huang, Y., An, T., Zhang, C., 2015. Pollution characteristics and health risk assessment of volatile organic compounds emitted from different plastic solid waste recycling workshops. Environ. Int. 77, 85–94.

Herrera, M., Matuschek, G., Kettrup, A., 2002. Thermal degradation of thermoplastic polyurethane elastomers (TPU) based on MDI. Polym. Degrad. Stabil. 78, 323–331.

Hoa, N.T.Q., Anh, H.Q., Tue, N.M., Trung, N.T., Da, L.N., Van Quy, T., Huong, N.T.A., Suzuki, G., Takahashi, S., Tanabe, S., Thuy, P.C., Dau, P.T., Viet, P.H., Tuyen, L.H., 2020. Soil and sediment contamination by unsubstituted and methylated polycyclic aromatic hydrocarbons in an informal e-waste recycling area, northern Vietnam: Occurrence, source apportionment, and risk assessment. Sci. Total Environ. 709, 135852.

Horii, Y., Ok, G., Ohura, T., Kannan, K., 2008. Occurrence and profiles of chlorinated and brominated polycyclic aromatic hydrocarbons in waste incinerators. Environ. Sci. Tech. 42, 1904–1909.

Huang, D.Y., Liu, C.P., Li, F.B., Liu, T.X., Liu, C.S., Tao, L., Wang, Y., 2014. Profiles, sources, and transport of polycyclic aromatic hydrocarbons in soils affected by electronic waste recycling in Longtang, south China. Environ. Monit. Assess. 186, 3351–3364.

Huang, H., Cai, C., Yu, S., Li, X., Liu, Y., Tao, S., Liu, W., 2019. Emission behaviors of nitro- and oxy-polycyclic aromatic hydrocarbons during pyrolytic disposal of electronic wastes. Chemosphere 222, 267–274.

Huang, X., Xu, X., Dai, Y., Cheng, Z., Zheng, X., Huo, X., 2020. Association of prenatal exposure to PAHs with anti-Mullerian hormone (AMH) levels and birth outcomes of newborns. Sci. Total Environ. 723, 138009.

Huo, X., Wu, Y., Xu, L., Zeng, X., Qin, Q., Xu, X., 2019. Maternal urinary metabolites of PAHs and its association with adverse birth outcomes in an intensive e-waste recycling area. Environ. Pollut. 245, 453–461.

Jiang, L.F., Cheng, Z.N., Zhang, D.Y., Song, M.K., Wang, Y.J., Luo, C.L., Yin, H., Li, J., Zhang, G., 2017. The influence of e-waste recycling on the molecular ecological network of soil microbial communities in Pakistan and China. Environ. Pollut. 231, 173–181.

Jin, R., Liu, G.R., Zheng, M.H., Jiang, X.X., Zhao, Y.Y., Yang, L.L., Wu, X.L., Xu, Y., 2017. Secondary copper smelters as sources of chlorinated and brominated polycyclic aromatic hydrocarbons. Environ. Sci. Tech. 51, 7945–7953.

Lee, H., Wang, L., Shih, J.F., 1995. Mutagenicity of particulates from the laboratory combustion of plastics. Mutat. Res. Lett. 346, 135–144.

Li, C.T., Zhuang, H.K., Hsieh, L.T., Lee, W.J., Tsao, M.C., 2001. PAH emission from the incineration of three plastic wastes. Environ. Int. 27, 61–67.

Li, Z., Romanoff, L.C., Lewin, M.D., Porter, E.N., Trinidad, D.A., Needham, L.L., Patterson, D.G., Sjödin, A., 2010. Variability of urinary concentrations of polycyclic aromatic hydrocarbon metabolite in general population and comparison of spot, first-morning, and 24-h void sampling. J. Expo. Sci. Environ. Epidemiol. 20, 526–535.

Lin, M., Tang, J., Ma, S., Yu, Y., Li, G., Mai, B., Fan, R., An, T., 2019. Simultaneous determination of polybrominated diphenyl ethers, polycyclic aromatic hydrocarbons and their hydroxylated metabolites in human hair: a potential methodology to distinguish external from internal exposure. Analyst 144, 7227–7235.

Lin, M., Tang, J., Ma, S., Yu, Y., Li, G., Fan, R., Mai, B., An, T., 2020b. Insights into biomonitoring of human exposure to polycyclic aromatic hydrocarbons with hair analysis: a case study in e-waste recycling area. Environ. Int. 136, 105432.

Lin, S.Y., Man, Y.B., Chow, K.L., Zheng, C.M., Wong, M.H., 2020a. Impacts of the influx of e-waste into Hong Kong after China has tightened up entry regulations. Crit. Rev. Environ. Sci. Tech. 50, 105–134.

Liu, F., Liao, C.Y., Fu, J.J., Lv, J.G., Xue, Q.Z., Jiang, G.B., 2014. Polycyclic aromatic hydrocarbons and organochlorine pesticides in rice hull from a typical e-waste recycling area in southeast China: temporal trend, source, and exposure assessment. Environ. Geochem. Health 36, 65–77.

Liu, J., Chen, X., Shu, H.Y., Lin, X.R., Zhou, Q.X., Bramryd, T., Shu, W.S., Huang, L.N., 2018. Microbial community structure and function in sediments from e-waste contaminated rivers at Guiyu area of China. Environ. Pollut. 235, 171–179.

Liu, R., Ma, S., Yu, Y., Li, G., Yu, Y., An, T., 2020. Field study of PAHs with their derivatives emitted from e-waste dismantling processes and their comprehensive human exposure implications. Environ. Int. 144, 106059.

Lopez, B.N., Man, Y.B., Zhao, Y.G., Zheng, J.S., Leung, A.O.W., Yao, J., Wong, M.H., 2011. Major pollutants in soils of abandoned agricultural land contaminated by e-waste activities in Hong Kong. Arch. Environ. Contam. Toxicol. 61, 101–114.

Lu, S.Y., Li, Y.X., Zhang, J.Q., Zhang, T., Liu, G.H., Huang, M.Z., Li, X., Ruan, J.J., Kannan, K., Qiu, R.L., 2016. Associations between polycyclic aromatic hydrocarbon (PAH) exposure and oxidative stress in people living near e-waste recycling facilities in China. Environ. Int. 94, 161–169.

Luo, J., Qi, S.H., Xie, X.M., Gu, X.W.S., Wang, J.J., 2017. The assessment of source attribution of soil pollution in a typical e-waste recycling town and its surrounding regions using the combined organic and inorganic dataset. Environ. Sci. Pollut. Res. 24, 3131–3141.

Luo, P., Bao, L.J., Li, S.M., Zeng, E.Y., 2015. Size-dependent distribution and inhalation cancer risk of particle-bound polycyclic aromatic hydrocarbons at a typical e-waste recycling and an urban site. Environ. Pollut. 200, 10–15.

Ma, J., Horii, Y., Cheng, J., Wang, W., Wu, Q., Ohura, T., Kannan, K., 2009. Chlorinated and parent polycyclic aromatic hydrocarbons in environmental samples from an electronic waste recycling facility and a chemical industrial complex in China. Environ. Sci. Technol. 43, 643–649.

Ma, S.T., Zeng, Z.H., Lin, M.Q., Tang, J., Yang, Y., Yu, Y.X., Li, G.Y., An, T.C., 2021. PAHs and their hydroxylated metabolites in the human fingernails from e-waste dismantlers: Implications for human non-invasive biomonitoring and exposure. Environ. Pollut. 283, 117059.

Molto, J., Egea, S., Conesa, J.A., Font, R., 2011. Thermal decomposition of electronic wastes: Mobile phone case and other parts. Waste Manag. 31, 2546–2552.

Nishimura, C., Horii, Y., Tanaka, S., Asante, K.A., Ballesteros, F.Jr, Pham Hung, V., Itai, T., Takigami, H., Tanabe, S., Fujimori, T., 2017. Occurrence, profiles, and toxic equivalents of chlorinated and brominated polycyclic aromatic hydrocarbons in e-waste open burning soils. Environ. Pollut. 225, 252–260.

Panagiotou, T., Levendis, Y.A., Carlson, J., Dunayevskiy, Y.M., Vouros, P., 1996. Aromatic hydrocarbon emissions from burning poly(styrene), poly(ethylene) and PVC particles at high temperatures. Combust. Sci. Technol. 116–117, 91–128.

Pang, J.X., Sun, B.Q., Li, H.Z., Mehler, W.T., You, J., 2012. Influence of bioturbation on bioavailability and toxicity of PAHs in sediment from an electronic waste recycling site in South China. Ecotoxicol. Environ. Saf. 84, 227–233.

Perkins, D.N., Brune Drisse, M.N., Nxele, T., Sly, P.D., 2014. E-waste: a global hazard. Ann. Glob. Health 80, 286–295.

Shen, C.F., Chen, Y.X., Huang, S.B., Wang, Z.J., Yu, C.N., Qiao, M., Xu, Y.P., Setty, K., Zhang, J.Y., Zhu, Y.F., Lin, Q., 2009. Dioxin-like compounds in agricultural soils near e-waste recycling sites from Taizhou area, China: chemical and bioanalytical characterization. Environ. Int. 35, 50–55.

Shen, H., Huang, Y., Wang, R., Zhu, D., Li, W., Shen, G., Wang, B., Zhang, Y., Chen, Y., Lu, Y., Chen, H., Li, T., Sun, K., Li, B., Liu, W., Liu, J., Tao, S., 2013. Global atmospheric emissions of polycyclic aromatic hydrocarbons from 1960 to 2008 and future predictions. Environ. Sci. Tech. 47, 6415–6424.

Shi, J., Zheng, G.J.S., Wong, M.H., Liang, H., Li, Y., Wu, Y., Li, P., Liu, W., 2016. Health risks of polycyclic aromatic hydrocarbons via fish consumption in Haimen bay (China), downstream of an e-waste recycling site (Guiyu). Environ. Res. 147, 233–240.

Simoneit, B.R.T., 2015. Triphenylbenzene in urban atmospheres, a new PAH source tracer. Polycycl. Aromat. Compd. 35, 3–15.

Simoneit, B.R.T., Medeiros, P.M., Didyk, B.M., 2005. Combustion products of plastics as indicators for refuse burning in the atmosphere. Environ. Sci. Tech. 39, 6961–6970.

Soler, A., Conesa, J.A., Ortuno, N., 2017. Emissions of brominated compounds and polycyclic aromatic hydrocarbons during pyrolysis of E-waste debrominated in subcritical water. Chemosphere 186, 167–176.

Song, Q.B., Li, J.H., 2014. A systematic review of the human body burden of e-waste exposure in China. Environ. Int. 68, 82–93.

Tang, J., Ma, S., Liu, R., Yue, C., Li, G., Yu, Y., Yang, Y., An, T., 2020. The pollution profiles and human exposure risks of chlorinated and brominated PAHs in indoor dusts from e-waste dismantling workshops: Comparison of GC-MS, GC-MS/MS and GC×GC-MS/MS determination methods. J. Hazard. Mater. 394, 122573.

Tang, X.J., Shen, C.F., Chen, L., Xiao, X., Wu, J.Y., Khan, M.I., Dou, C.M., Chen, Y.X., 2010. Inorganic and organic pollution in agricultural soil from an emerging e-waste recycling town in Taizhou area. China J. Soil. Sediment. 10, 895–906.

Tomsej, T., Horak, J., Tomsejova, S., Krpec, K., Klanova, J., Dej, M., Hopan, F., 2018. The impact of co-combustion of polyethylene plastics and wood in a small residential boiler on emissions of gaseous pollutants, particulate matter, PAHs and 1,3,5-triphenylbenzene. Chemosphere 196, 18–24.

Tue, N.M., Goto, A., Takahashi, S., Itai, T., Asante, K.A., Nomiyama, K., Tanabe, S., Kunisue, T., 2017. Soil contamination by halogenated polycyclic aromatic hydrocarbons from open burning of e-waste in Agbogbloshie (Accra, Ghana). J. Mater. Cycles Waste Manag. 19, 1324–1332.

Valavanidis, A., Iopoulos, N., Gotsis, G., Fiotakis, K., 2008. Persistent free radicals, heavy metals and PAHs generated in particulate soot emissions and residue ash from controlled combustion of common types of plastic. J. Hazard. Mater. 156, 277–284.

Wang, J., Chen, S., Tian, M., Zheng, X., Gonzales, L., Ohura, T., Mai, B., Simonich, S.L.M., 2012a. Inhalation cancer risk associated with exposure to complex polycyclic aromatic hydrocarbon mixtures in an electronic waste and urban area in south China. Environ. Sci. Tech. 46, 9745–9752.

Wang, Q., Miyake, Y., Amagai, T., Suzuki, G., Matsukami, H., Tue, N.M., Takahashi, S., Tanabe, S., Tuyen, L.H., Viet, P.H., Takigami, H., 2016. Halogenated polycyclic aromatic hydrocarbons in soil and river sediment from e-waste recycling sites in Vietnam. J. Water Environ. Technol. 14, 166–176.

Wang, Y., Tian, Z., Zhu, H., Cheng, Z., Kang, M., Luo, C., Li, J., Zhang, G., 2012b. Polycyclic aromatic hydrocarbons (PAHs) in soils and vegetation near an e-waste recycling site in South China: Concentration, distribution, source, and risk assessment. Sci. Total Environ. 439, 187–193.

Wang, Y., He, J., Wang, S., Luo, C., Yin, H., Zhang, G., 2017. Characterisation and risk assessment of polycyclic aromatic hydrocarbons (PAHs) in soils and plants around e-waste dismantling sites in southern China. Environ. Sci. Pollut. Res. 24, 22173–22182.

Wei, S., Liu, M., Huang, B., Bi, X., Sheng, G., Fu, J., 2011. Polycyclic aromatic hydrocarbons with molecular weight 302 in PM2.5 at two industrial sites in South China. J. Environ. Monit. 13, 2568–2574.

Wei, S., Huang, B., Liu, M., Bi, X., Ren, Z., Sheng, G., Fu, J., 2012. Characterization of PM2.5-bound nitrated and oxygenated PAHs in two industrial sites of South China. Atmos. Res. 109–110, 76–83.

Wen, S., Yang, F.X., Gong, Y., Zhang, X.L., Hui, Y., Li, J.G., Liu, A.L., Wu, Y.N., Lu, W.Q., Xu, Y., 2008. Elevated levels of urinary 8-hydroxy-2′-deoxyguanosine in male electrical and electronic equipment dismantling workers exposed to high concentrations of polychlorinated dibenzo-p-dioxins and dibenzofurans, polybrominated diphenyl ethers, and polychlorinated biphenyls. Environ. Sci. Technol. 42, 4202–4207.

Wheatley, L., Levendis, Y.A., Vouros, P., 1993. Exploratory study on the combustion and PAH emissions of selected municipal waste plastics. Environ. Sci. Tech. 27, 2885–2895.

Wong, M.H., Wu, S.C., Deng, W.J., Yu, X.Z., Luo, Q., Leung, A.O.W., Wong, C.S.C., Luksemburg, W.J., Wong, A.S., 2007. Export of toxic chemicals - a review of the case of uncontrolled electronic-waste recycling. Environ. Pollut. 149, 131–140.

Xu, P.J., Tao, B., Ye, Z.Q., Zhao, H., Ren, Y., Zhang, T., Huang, Y.R., Chen, J.P., 2016. Polycyclic aromatic hydrocarbon concentrations, compositions, sources, and associated carcinogenic risks to humans in farmland soils and riverine sediments from Guiyu, China. J. Environ. Sci. 48, 102–111.

Xu, X., Yekeen, T.A., Xiao, Q., Wang, Y., Lu, F., Huo, X., 2013. Placental IGF-1 and IGFBP-3 expression correlate with umbilical cord blood PAH and PBDE levels from prenatal exposure to electronic waste. Environ. Pollut. 182, 63–69.

Xu, X., Liu, J., Huang, C., Lu, F., Chiung, Y.M., Huo, X., 2015. Association of polycyclic aromatic hydrocarbons (PAHs) and lead co-exposure with child physical growth and development in an e-waste recycling town. Chemosphere 139, 295–302.

Yu, X.Z., Gao, Y., Wu, S.C., Zhang, H.B., Cheung, K.C., Wong, M.H., 2006. Distribution of polycyclic aromatic hydrocarbons in soils at Guiyu area of China, affected by recycling of electronic waste using primitive technologies. Chemosphere 65, 1500–1509.

Zeng, Z., Huo, X., Wang, Q., Wang, C., Hylkema, M.N., Xu, X., 2020. PM2.5-bound PAHs exposure linked with low plasma insulin-like growth factor 1 levels and reduced child height. Environ. Int. 138 (105660).

Zhang, D.L., An, T.C., Qiao, M., Loganathan, B.G., Zeng, X.Y., Sheng, G.Y., Fu, J.M., 2011. Source identification and health risk of polycyclic aromatic hydrocarbons associated with electronic dismantling in Guiyu town, South China. J. Hazard. Mater. 192, 1–7.

Zhang, K., Schnoor, J.L., Zeng, E.Y., 2012. E-waste recycling: where does it go from here? Environ. Sci. Tech. 46, 10861–10867.

Zheng, X., Huo, X., Zhang, Y., Wang, Q., Zhang, Y., Xu, X., 2019. Cardiovascular endothelial inflammation by chronic coexposure to lead (Pb) and polycyclic aromatic hydrocarbons from preschool children in an e-waste recycling area. Environ. Pollut. 246, 587–596.