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Heavy metals in indoor dust: Spatial distribution, influencing factors, and potential health risks

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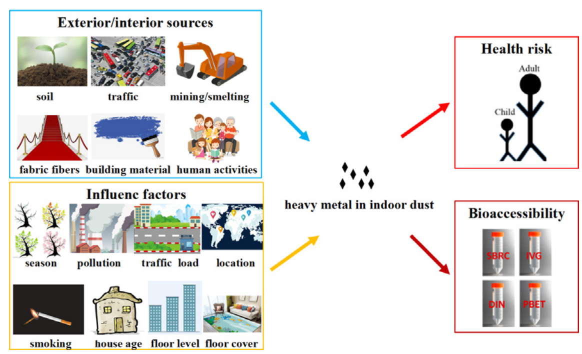
HIGHLIGHTS

- The global status of heavy metal accumulation in indoor dust was reviewed.
- Control of Cd and Pb pollution in indoor dust should be given priority.
- Dust Pb exposure is a major health concern in e-waste recycling areas.
- Children exposure to Pb via indoor dust in developing countries should not be overlooked.

GRAPHICAL ABSTRACT



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ABSTRACT

Given the large proportion of time that people spend indoors, the potential health risks posed by heavy metals in the indoor environment deserve greater attention. A global-scale assessment of heavy metal contamination in indoor dust was conducted in this study based on >127 articles published between 1985 and 2019. The pollution levels, spatio-temporal variations, sources, bioaccessibilities, influencing factors, and health risks of heavy metals associated with indoor dust were analyzed. Children's blood lead levels (BLLs) were also estimated using the integrated exposure uptake biokinetic model. The results indicated that the median concentrations of Cu and Zn in 71.9% and 71.0% of the study sites surpassed the corresponding permissible limits, 100 and 300 mg/kg, respectively; thus, their control should be given priority. Heavy metal concentrations in indoor dust from different areas of the world varied greatly, which was closely associated with the type of local human activities, such as mining, melting, e-waste recycling and Pb-related industries. The bioaccessibilities of some key elements, e.g., Pb, Cd, Cu, and Zn, in household dust were high. The levels of heavy metals in indoor dust were mainly affected by a combination of outdoor and indoor sources and related critical factors, and future studies should focus on quantifying the contributions of different sources. Based on the health risk assessment, dust Pb exposure is a major health concern in e-waste recycling areas, which warrants greater attention. 49.8%, 36.8% and 14.4% of study sites showed BLLs exceeding 35 μg/L (threshold limit in Germany), 50 μg/L (threshold limit in the USA), or 100 μg/L (threshold limit in China), respectively. Finally, Pb exposure from indoor dust represents a major contributor to children's blood Pb poisoning in many developing countries. This study details the overall heavy metal contamination status of indoor dust and provides insights for policymakers with respect to pollution prevention measures.

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1. Introduction

As humans become increasingly urban, the indoor environment is gaining increasing attention worldwide. People spend >90% of their time exposed to the indoor atmosphere (Andrade and Dominski, 2018; US EPA, 1989). Concentrations of many metals and metalloids are higher in indoor dust than in outdoor settled dust or soil in ordinary urban environments (Davies et al., 1987). Longer durations and elevated contaminant concentrations in the indoor environment may increase the chances of exposure to some contaminants by 1000 fold compared with those of outdoor exposure (Hwang et al., 2008). Indoor air pollution is a serious health problem, as it causes approximately 4.5 million annual deaths globally owing to pneumonia (12%), stroke (34%), ischemic heart diseases (26%), chronic obstructive pulmonary diseases (22%), and lung cancer (6%) (WHO, 2018).

Prominent among the wide variety of contaminants reported as present in the indoor environment, heavy metals are of concern owing to their non-degradability, high toxicity, and adverse effects on humans. Many of these contaminants adsorb to particulate matter suspended in indoor air, which later settles as house dust. Heavy metals in dust can accumulate in human fatty tissues and internal organs via direct inhalation, ingestion, and dermal contact absorption, thereby posing risks to human health, especially for young children (Madany et al., 1987). Incidental ingestion of house dust is the predominant contributor to Pb in children's blood (Li et al., 2015). The US EPA has ranked reducing childhood Pb exposure as a high priority and strengthened the dust Pb hazard standards from 40 $\mu g/ft^2$ and 250 $\mu g/ft^2$ to 10 $\mu g/ft^2$ and 100 $\mu g/ft^2$ on floors and window sills, respectively, in 2018 (US EPA, 2018).

In recent years, several review articles have provided assessments on heavy metal pollution in indoor dust (Amoatey et al., 2018; Kastury et al., 2017; Obeng-Gyasi, 2019; Tan et al., 2016; Li, 2015). However, the scope of these studies was relatively narrow, as they usually focused on a single aspect, such as bioaccessibility (Ibanez et al., 2010; Kastury et al., 2017; Turner, 2011), health risk (Tan et al., 2016), mutagenic hazards (Maertens et al., 2004), sources and speciation (Fergusson and Kim, 1991) or on a single element (Obeng-Gyasi, 2019), or in a single region (Li, 2015). Indoor dust pollution was also assessed as a part of a few review articles on indoor air quality (Amoatey et al., 2018; Charlesworth et al., 2011; Tham, 2016), but was not their focus. Here, the literature is utilized to determine the status of studies of heavy metals in indoor dust; to determine the differences in the concentrations and distribution characteristics of heavy metals in indoor dust across broad geographical regions; to assess the bioaccessibility, sources, and influencing factors; and to evaluate the potential health risks to adults and children. Overall, this study provides multi-factor evaluation on exposure situations regarding to heavy metals in indoor dust across broad geographical regions.

2. Methods

2.1. Data sources and extraction

Here we retrieved a total of 196 articles corresponding to the keywords indoor dust, settled dust, and heavy metal from databases such as the Web of Science, Elsevier Science Direct, and Science Online. The following eligibility criteria were applied to the literature (Table S1). Dust heavy metal concentrations and dust loading rates are different measures of indoor environmental heavy metal levels. Dust heavy metal studies reporting heavy metal loading rate were excluded in this study (Khoder et al., 2010; Meyer et al., 1999a; Moriske et al., 1996). The literature with heavy metal concentrations that were bioavailable (Tan et al., 2018), measured in $\mu g/m^3$ (Ruggieri et al., 2019; Taner et al., 2013), or not clearly defined (Soto-Jiménez and Flegal, 2011; Junaid et al., 2017a; Junaid et al., 2017b) was excluded. In addition, the literature with dust samples collected from both indoor and outdoor areas without subdivision was excluded (Bi et al., 2015; Chen et al., 2014). The specific literature selection flow is shown in Fig. S1. A table (Table S4) summarizing all 127 articles is shown in the supplementary materials. The amount and location of sampling sites, the type and concentrations of heavy metals, latitude-longitude coordinate, sampling time, the reference title, etc. are include in the table.

The data on heavy metal concentrations in indoor dust were collected from the selected literature, and the unit of heavy metal concentration was standardized as mg/kg. The sampling and processing methods used in these studies are widely accepted by the scientific community. Dust samples were collected by vacuuming (Kelepertzis et al., 2019; Rasmussen et al., 2011; Lanzerstorfer, 2017) or sweeping a soft plastic brush into a plastic dustpan (Zhou et al., 2019; Iwegbue et al., 2018; Iwegbue et al., 2018; Lucas et al., 2015) and packing the samples into acid-washed polyethylene bags in most of the studies. Across the studies, various methods were used to measure the heavy metal concentrations in the dust samples, but atomic absorption spectroscopy and inductively coupled plasma-mass spectrometry were the most commonly used.

2.2. Study environment

Indoor dust pollution of heavy metals has been extensively studied in schools (Liu et al., 2016; Habil et al., 2013; Zhong et al., 2014; Sulaiman et al., 2017), especially preschools (Latif et al., 2014; Lu et al., 2014) and primary school (Akinwunmi et al., 2017; Praveena et al., 2015). Residential homes (Olujimi et al., 2015; Bi et al., 2015; Tong, 1998) and offices (Jaradat et al., 2004; Iwegbue et al., 2018) were also the focus of the scope of this research. Compared with that of the residential homes, offices showed a higher degree of dustiness and heavy metals (Lisiewicz et al., 2000). Public places, including internet cafes (Iwegbue et al., 2018) and subway stations (Kim et al., 1998), were also investigated. The levels of Cu and Pb in indoor dust from subway stations in the city center were high (Kim et al., 1998). Heavily polluted areas, including work places in industrial estates (Al-Khashman, 2004), paint manufacturing plants (Huang et al., 2010), electronic and electrical material maintenance shops (Getachew et al., 2019), and electronic workshops (Deng et al., 2014; Iwegbue et al., 2018; Xu et al., 2015), were also research hotspots. Numerous studies have reported the associated risks that may arise from human exposure to heavy metals in dust from these heavily polluted areas (Bi et al., 2011; Iwegbue et al., 2018). Other studies on indoor dust in some microenvironments are sporadic, such as in natural history museums (Marcotte, 2017), mosque (El-Mubarak et al., 2016) and buses (Gao et al., 2015; Lei et al., 2016).

2.3. Sample location

Based on the selected literature, indoor dust samples were collected from various surfaces of houses (Bi et al., 2015; Cao et al., 2016), such as stairs, entryways, doorsteps (Lu et al., 2014), balconies, floors (Jelenska et al., 2017; Johnson et al., 2005; Praveena et al., 2015), windows (Cao et al., 2016; Lu et al., 2014; Praveena et al., 2015), fans, and corners (Cao et al., 2016). Indoor dust was also collected from air conditioner filters (Hu et al., 2018; Huang et al., 2014b; Siddique et al., 2011) or vacuum cleaner bags (Seifert et al., 2000). The heavy metal concentrations in indoor dust varied greatly among the sampling locations of a given house. The highest Pb concentration in dust was noted in samples from under a doormat (Davies et al., 1987). Likewise, the highest average concentrations of Pb, Ni, Cd, Co, Cu, and Cr were found in an entryway (Hassan, 2012). This may be because the doormat and footsteps are the most likely locations for the removal of external dust (Davies et al., 1987; Tan et al., 2016). The average heavy metal concentrations of Pb, Cu, and Cd were higher in dust on windows than that on floors and fans when fans and open windows were used as ventilation system (Praveena et al., 2015). In addition to that in common indoor environments, dust collected from industries is of high scientific and policy interest. For example, dust from electronic waste recycling facilities has been the focus of many investigations (Deng et al., 2014; Getachew et al., 2019; Yu et al., 2019).

2.4. Elements selected for this study

Based on data extracted from the 127 articles (Table S4), studies investigating exposure to toxic contaminants in indoor dust have often focused on Pb, followed by Zn, Cu, Cd, Cr, Ni and As. Thus, the contamination of these elements in indoor dust is summarized here according to the quantity and quality of the presented data in the references. Unlike other heavy metals that tend to exist as suspended particulate species in the atmosphere, Hg exists mainly as gas. Nevertheless, as Hg in indoor dust is also a matter of concern under certain conditions (Bavec et al., 2017; Huang et al., 2014b; Lin et al., 2017; Marcotte, 2017; Rasmussen et al., 2011; Yang et al., 2015), it is also included in this study.

Other trace elements have also been occasionally studied. Such as Co (Bavec et al., 2017; Beamer et al., 2012; Cao et al., 2016; El-Mubarak et al., 2016; Fergusson et al., 1986; Hu et al., 2018; Li et al., 2020; Lin et al., 2015; Nasir et al., 2015; Iga and Beata, 2016; Xiang et al., 2016), V (Beamer et al., 2012; Cao et al., 2016; El-Mubarak et al., 2016; Fergusson et al., 1986; Hu et al., 2018; Wan et al., 2016), Sb (Beamer et al., 2012; Hu et al., 2018; Huang et al., 2014a; Li et al., 2016; Xiang et al., 2016), Se (Ali et al., 2019; Cao et al., 2016), Ti (Ali et al., 2019; Beamer et al., 2012; Hu et al., 2018; Rasmussen et al., 2001), Ba (Beamer et al., 2012; El-Mubarak et al., 2016; Hu et al., 2018; Rasmussen et al., 2001), Mo (Bavec et al., 2017; Beamer et al., 2012), U (El-Mubarak et al., 2016; Rasmussen et al., 2018), Ag (Beamer et al., 2012; Rasmussen et al., 2018), Sn (Ali et al., 2019; Beamer et al., 2012; Huang et al., 2014a), B (Beamer et al., 2012; Rasmussen et al., 2018), Li (Ali et al., 2019; El-Mubarak et al., 2016), Hf, Th, Sc, Sm, Ce, La (Fergusson et al., 1986), Cs, Sm, Ta, Th, Yb, Nd, Rb (El-Mubarak et al., 2016; Siddique et al., 2011), Sr (Hu et al., 2018), and Si (Nasir et al., 2015).

Moreover, major elements such as Fe (Beamer et al., 2012; El-Mubarak et al., 2016; Iga and Beata, 2016; Rasmussen et al., 2018; Wang et al., 2020), Al (Beamer et al., 2012; El-Mubarak et al., 2016; Nasir et al., 2015; Rasmussen et al., 2018), K (Beamer et al., 2012; El-Mubarak et al., 2016; Fergusson et al., 1986), Mn (Beamer et al., 2012; El-Mubarak et al., 2016; Huang et al., 2014a; Iga and Beata, 2016; Nasir et al., 2015; Rasmussen et al., 2018; Wang et al., 2020; Xiang et al., 2016) have also been investigated.

2.5. Characterization methods

The characterization of dust includes the pollution level of heavy metals, microscope observations, and chemical element analysis. Magnetic particles and heavy metals coexist in dust. The measurement of magnetic properties is a simple, rapid, sensitive, and non-destructive method, so magnetic methods have been successfully applied to evaluate the pollution level of dust (Zhu et al., 2012; Jelenska et al., 2017). The micro-morphological and mineralogical characteristics of indoor dust were determined. Conventional scanning electron microscopes combined with energy dispersive spectra and advanced microscopic techniques, i.e., focused ion-beam and electron back scatter diffraction, were applied in a number of studies to better characterize the morphology and geochemical composition of indoor dust particles (Jelenska et al., 2017; Torres-Sánchez et al., 2017; Zhou et al., 2019).

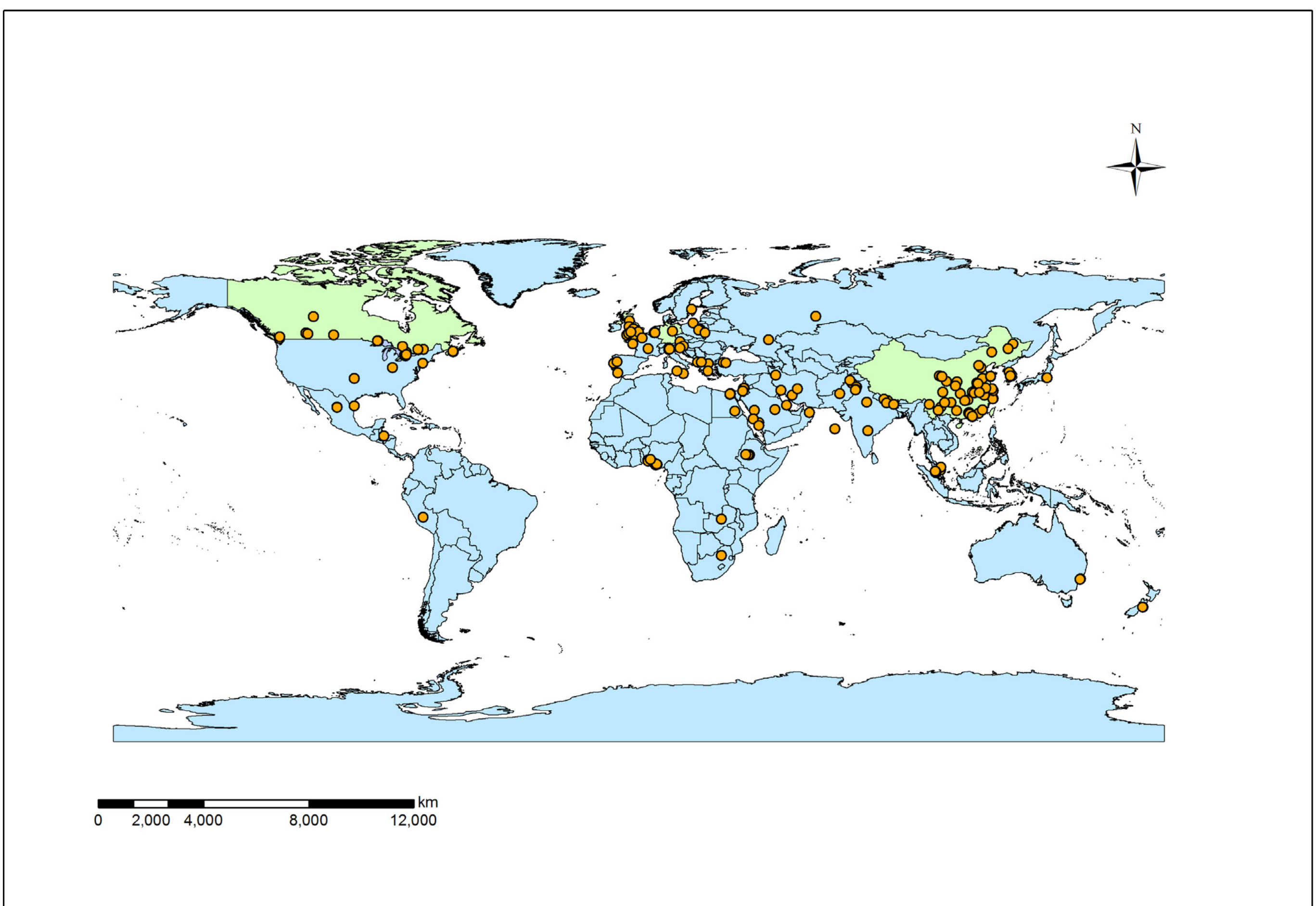


Fig. 1. Geographical location of indoor dust studies retrieved from the scientific literature.

3. Results and discussion

3.1. Distribution of examined sites

Data on indoor dust were obtained across 36 countries worldwide based on previous studies. Each of these studies investigated only one or a few sites that were suspected of being polluted. As shown in Fig. 1, the examined sites are densely distributed in European countries and China, but clearly sparse in some regions, such as Russia, Australia, and South America. It to some extent reflects that economically developed regions have been the focus of more intensive research on indoor dust pollution.

Indoor dust investigations have been conducted in many countries at the national scale. A national survey of heavy metals in dust was commissioned by the United Kingdom Department of the Environment, and sampling was conducted in 1981–1982 in 53 locations in England, Scotland, and Wales (Thornton et al., 1990). It was revealed that in 10% of the 5228 homes, the tested Pb concentration was in excess of 2000 mg/kg with a geometric mean value for Pb in house dust of 561 mg/kg (excluding old mining areas) (Thornton et al., 1985; Thornton et al., 1990). The Canadian House Dust Study was conducted in Canada in 2007–2010; it was designed to obtain statistically robust, nationally representative baseline estimates for Pb and other chemical constituents of settled dust based on nationally representative random samples (Rasmussen et al., 2011). The results of the investigation showed that the average bioaccessible Pb concentration in the <80 μm fraction of 1025 dust samples was 74 mg/kg in Canada (Rasmussen et al., 2011). The German Environmental Survey (GerES) is a large-scale, representative population study that has been conducted four times. The first survey was conducted in 1985–1986 followed by GerES II in 1990–1992, GerES III in 1998 and GerES IV in 2003–2006 (Seifert et al., 2000). Environmental samples including house dust were analyzed to characterize exposure in the domestic environment (Seifert et al., 2000). The median concentrations of As, Cd, Cr, Cu, Pb and Zn in vacuum cleaner bags were 2.1, 0.9, 63, 76, 4 and 469 mg/kg, respectively (Seifert et al., 2000). To investigate dust Pb contamination in rural households, 122 dust samples were collected from rural households in approximately one quarter of the provinces in China in 2009 (Yang et al., 2011). The mean Pb concentration in household dust was 207.5 mg/kg (Yang et al., 2011). The investigation of heavy metal pollution in indoor dust would benefit every country and region by obtaining default values to be incorporated into risk assessments. Therefore, further nationwide investigations in countries worldwide are needed.

3.2. Pollution levels and spatio-temporal variation

The concentrations of heavy metals in indoor dust worldwide are presented in Fig. 2. Currently, there are no guidelines (standard reference values) for heavy metals in dust. Therefore, the WHO environmental quality standards for soils were used as references to assess the contamination levels of heavy metals in dust. The median concentrations of As, Cd, Cr, Cu, Ni, Pb and Zn in 13.9%, 30.0%, 25.8%, 71.9%, 42.7%, 58.1% and 71.0% of the study sites surpassed the corresponding permissible limits (20, 3, 100, 100, 50, 100 and 300 mg/kg) respectively. Therefore, the risks of exposure to Cu and Zn are the highest in all considered heavy metals, followed by Pb and Ni.

To characterize the spatial distribution of heavy metals, GIS was used to construct contamination maps with likely hotspots. The heavy metal concentrations in indoor dust varied greatly among the geographic locations (Fig. 3). The distribution of hotspots indicated that mining and smelting activities, e-waste recycling, and industrial production significantly contribute to heavy metal accumulation in indoor dust. For example, the median Cd, Pb, and Zn concentrations in house dust from the mining village of Stratoni, Greece were 7.2, 990 and 1970 mg/kg, respectively (Argyraki, 2013). The Pb concentration in dust near a smelter in Yunnan, China ranged from 191 mg/kg to 13,371 mg/kg (Xie et al., 2013). Dust Zn concentrations were high in Bagnolo Mella (median 524 mg/kg), which was caused by a local active ferromanganese alloy plant (Lucas et al., 2015). The Cd concentration reached a maximum of 59 mg/kg in house dust from an e-waste area in South China (Zheng et al., 2013), which was approximately 20 times greater than the permissible limit (3 mg/kg) set by WHO standard. Likewise, high Cd (median 40.4 mg/kg), Cu (median 1200 mg/kg) and Pb (median 1380 mg/kg) levels in dust from Qingyuan and Guangzhou, China were also caused by e-waste recycling (He et al., 2017). The average amounts of Cd, Pb, Cu, Zn, and Cr in dust from electronic and electrical material maintenance workshops were found to be in the ranges of 9.62–12.30, 58,980–73,970, 5778–8570, 261.6–708.9, and 1127–1273 mg/kg, respectively, from Oromia, Ethiopia (Getachew

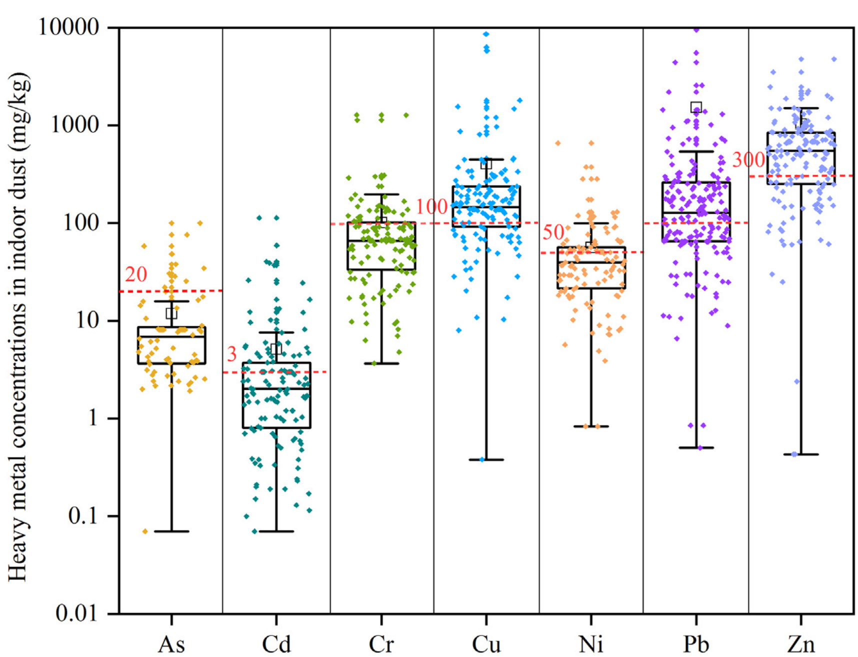


Fig. 2. Heavy metal concentrations in indoor dust.

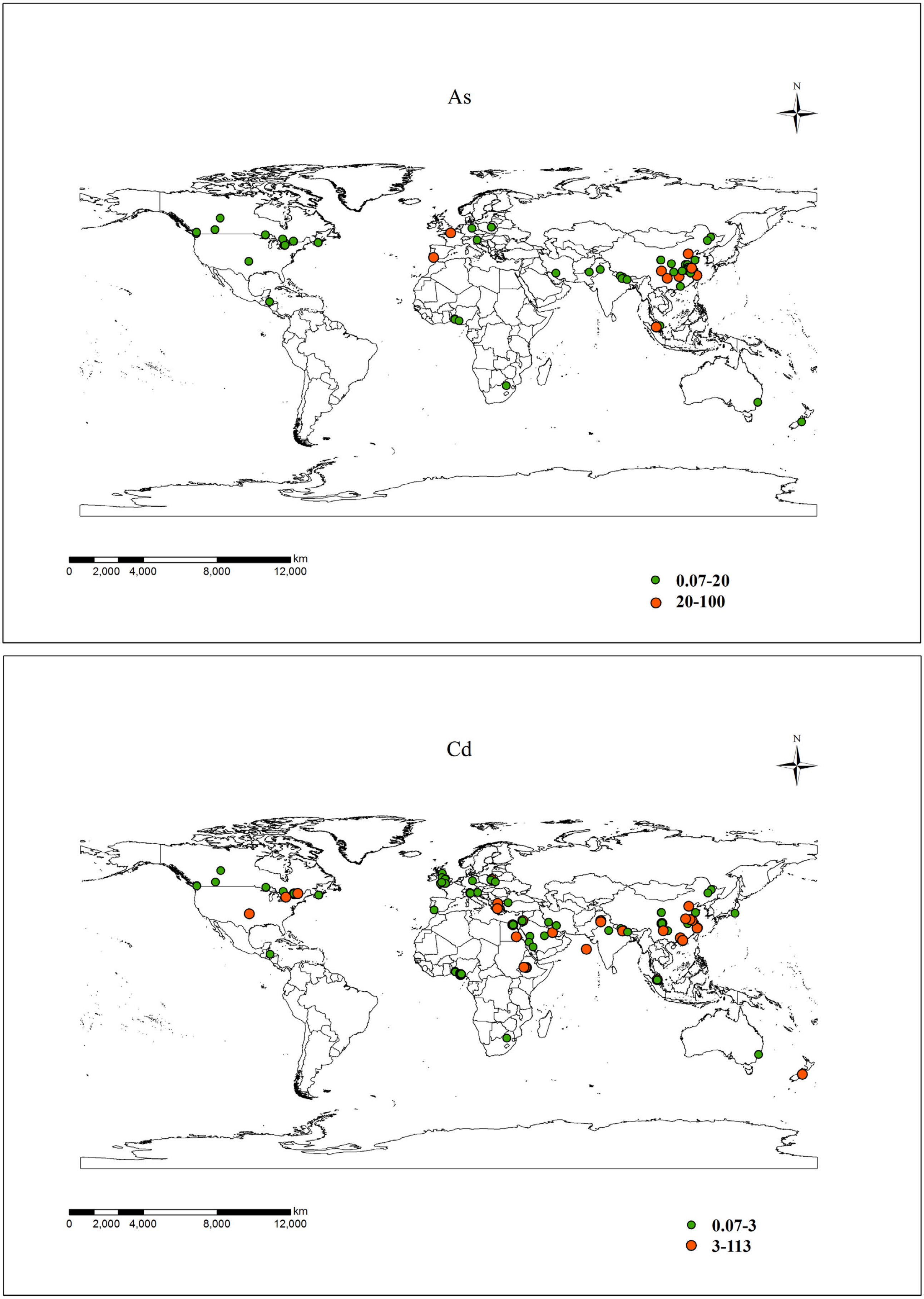
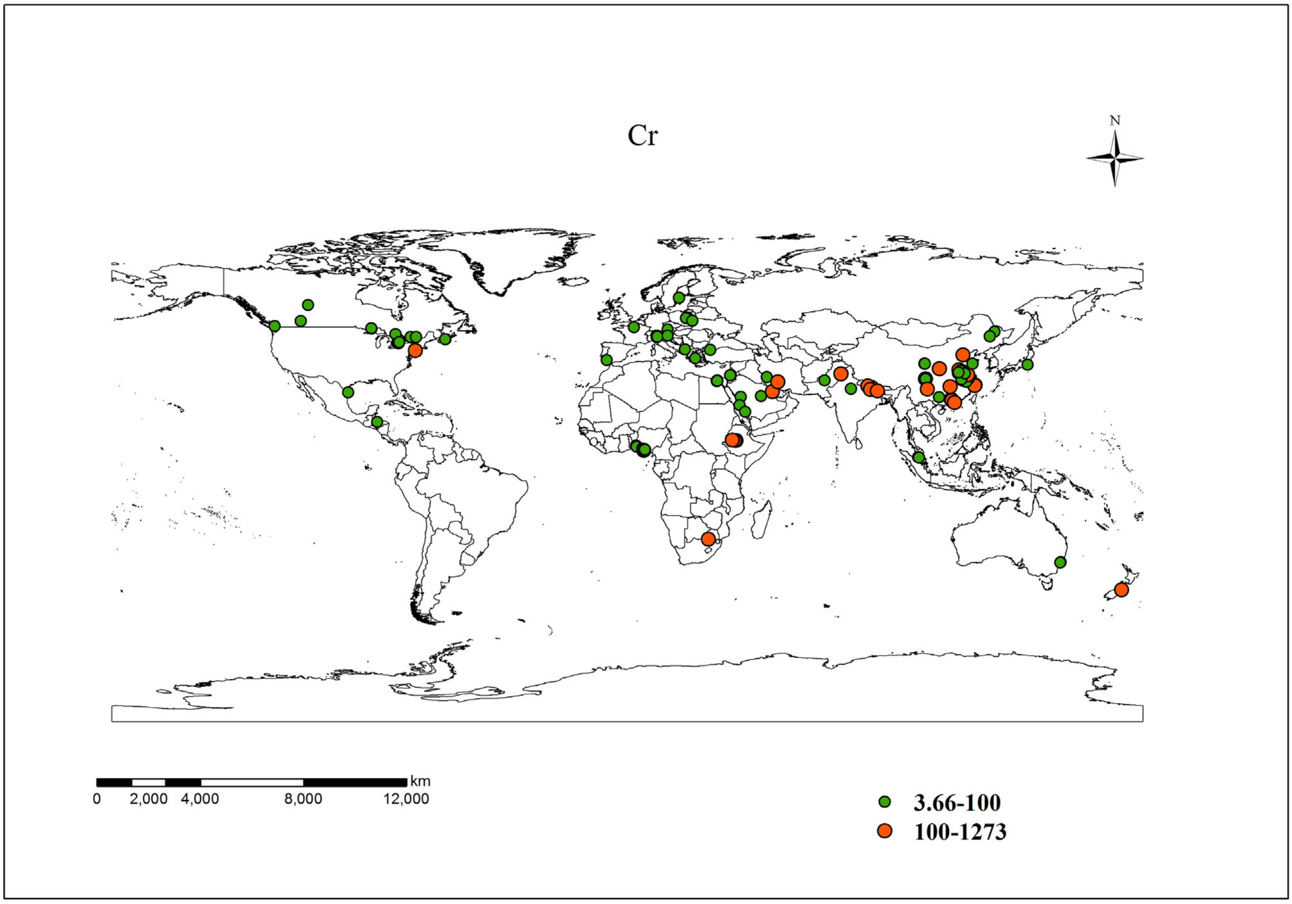
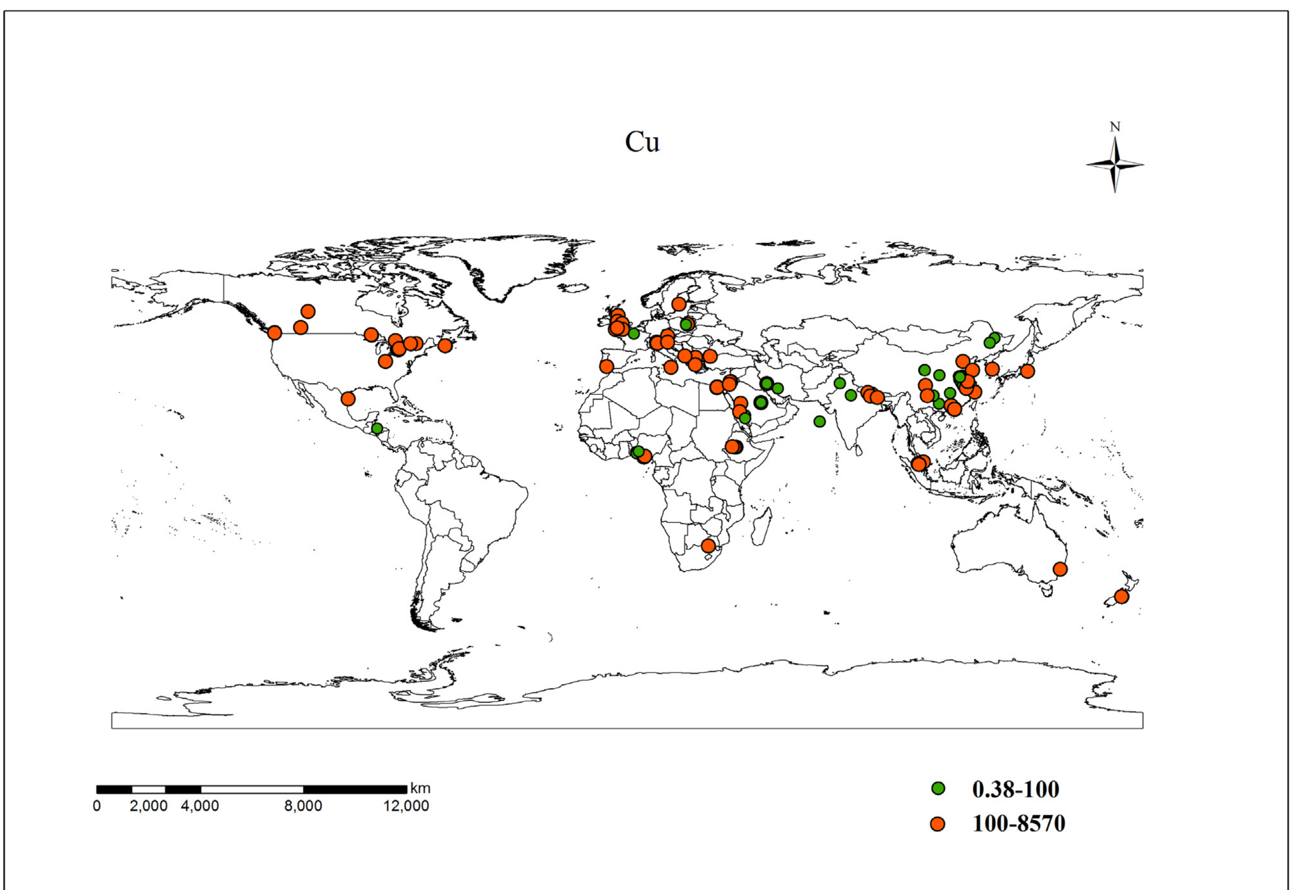


Fig. 3. Spatial distribution of heavy metal concentrations in indoor dust in different regions based on existing data.





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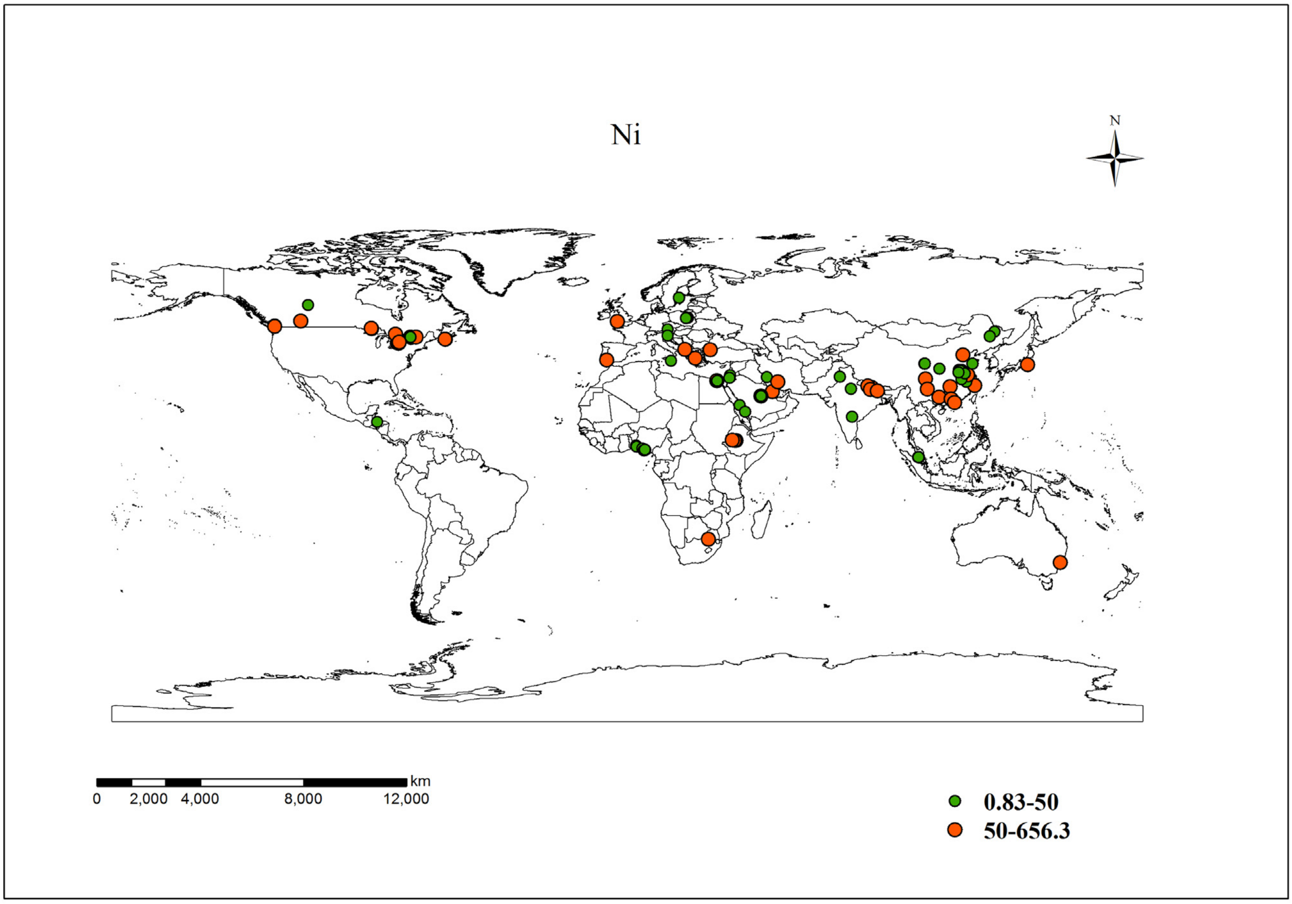

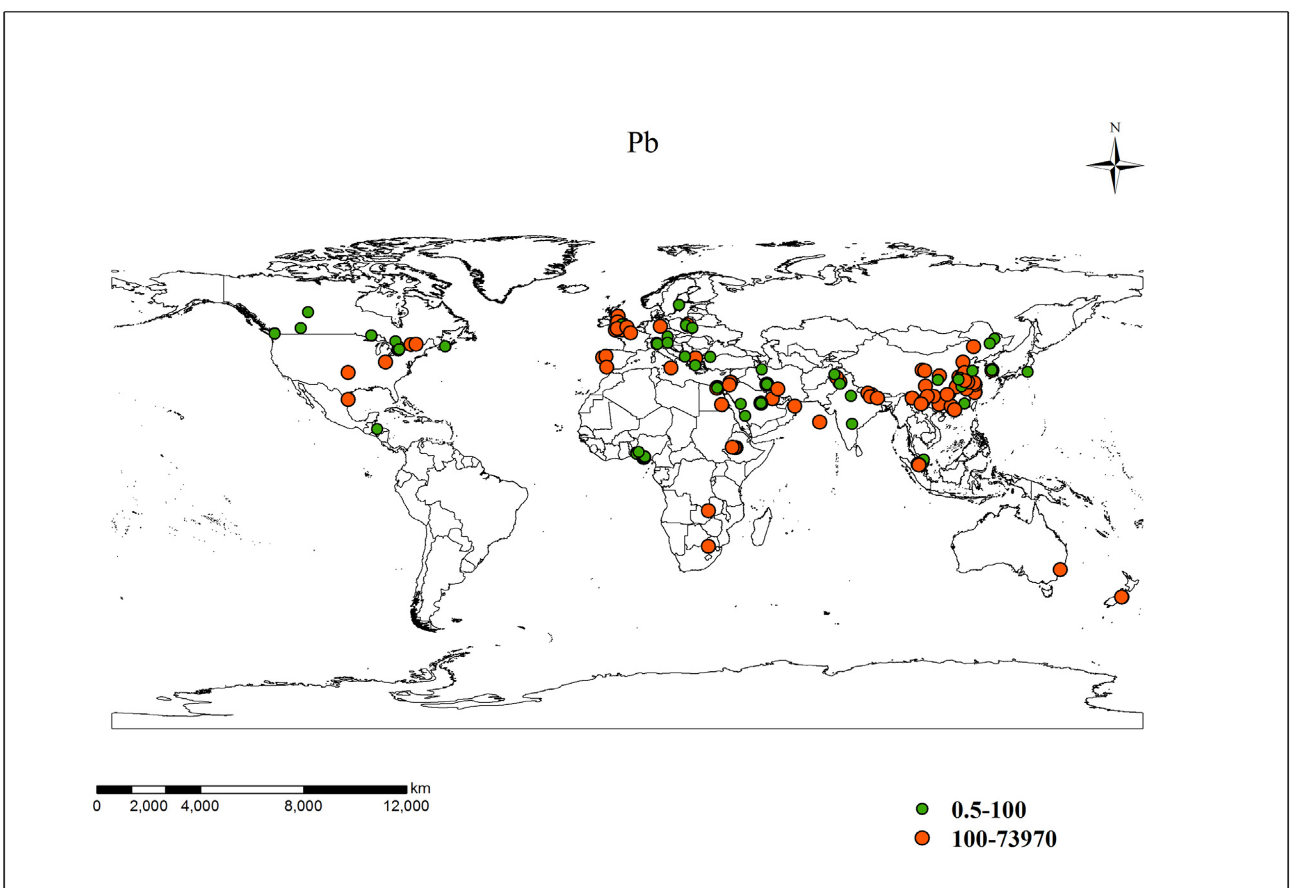

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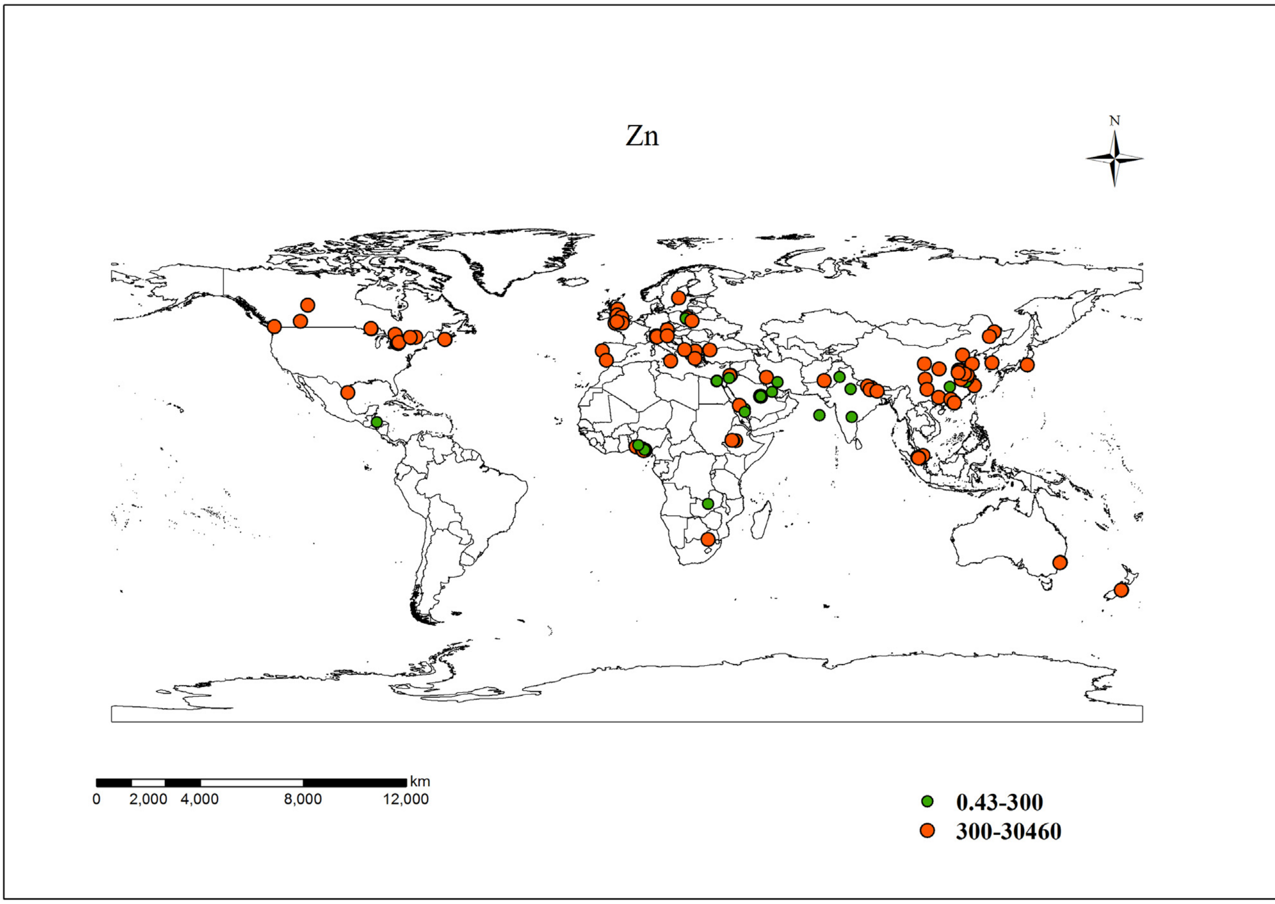

Fig. 3 (*continued*).

et al., 2019). The Pb concentrations in indoor dust from electronic workshops showed high levels (41.1–4810 mg/kg) (Iwegbue et al., 2018). The mean concentration of Pb in solvent-based paint dust from a paint manufacturing plant was found to be 15,680 mg/kg (Huang et al., 2010).

Based on national surveys in the United Kingdom, the geometric mean concentrations of Pb in household dust were 561 mg/kg in 1981–1982 (Thornton et al., 1990) and 150 mg/kg in 2005 (Turner and Simmonds, 2006), respectively. This significant temporal reduction in Pb concentrations in the UK household dust may be connected with the efforts to address air pollution by the UK government. In 1994, a call for research proposals on air pollution was published jointly by the Department of Health, the Department of the Environment, and the Medical Research Council (Davison and Hewitt, 1997). Each year since 1995, a review of the programme of air pollution, which has expanded and now includes work on indoor air pollutants, has been held. A revision of the United Kingdom national air quality strategy was published in early 1999 (Department of the Environment, Transport, and the Regions, 1999). In contrast, the geometric mean concentrations of Pb in household dust were 119 mg/kg in 1993 in Ottawa (Rasmussen et al., 2001) and 232.6 mg/kg in 2007–2010 based on national survey in Canada (Rasmussen et al., 2013), respectively. In other regions, such as China, USA, and Germany, temporal comparisons are not possible due to differences in sample sites and scales of the corresponding studies.

Nevertheless, it is worth noting that there are still flaws in the spatio-temporal patterns presented here. The format of the data from the collected papers was not uniform in different time intervals. Due to the limited amount of data, spatial distribution of heavy metal concentrations in indoor dust was not assessed in different time intervals, which may not reflect the actual situation. In addition, the temporal trend of heavy metal concentrations was extrapolated based on geometric mean, being neither median nor arithmetic mean. However, the temporal trend might somewhat depend on the statistic approach. Further verification of the temporal trends of heavy metal in indoor dust is therefore necessary.

3.3. Heavy metals in indoor dust from different microenvironments

The concentrations of heavy metals varied widely by element and by microenvironment (Fig. 4). The levels of contamination in e-waste workshops were the highest, the median concentrations of Pb, Cu, Ni and Cd were approximately 67, 63, 5.37 and 4.27 times greater than the corresponding permissible WHO limits for soils, respectively. Thus, there is an urgent need to mitigate the potentially severe toxic effects of e-waste recycling on the environment and humans. The median concentration of Pb (5507 mg/kg) in natural history museums was nearly 55 times higher than the permissible limit of WHO. This may be attributable to the use of Pb paint on stuffed specimens and walls of the museums (Marcotte, 2017). The median concentrations of Cd, Cr, As, Cu and Zn in industrial workplaces surpassed the permissible limits of WHO. Indoor dust from public places, residential houses and offices showed high concentrations of Zn, Cu and Pb. The color of wall paints is a significant source of heavy metals in indoor dust (Kurt-Karakus, 2012). Yellow paint is associated with very high levels of Cd, Cu, Pb, and Zn, purple paint is related to higher concentrations of Zn and Pb, and green paint is related to Cu (Tong and Lam, 2000). Moreover, the high concentrations could be caused by the fact that these heavy metals are derived from industrial emissions located close to industrial areas (Jaradat et al., 2004; Iwegbue et al., 2018). The median concentration of Zn in bus dust was slightly higher than the permissible limit of WHO. The settled dust in buses was mainly derived from the suspended particles in the aerial environment of the streets, which entered via the

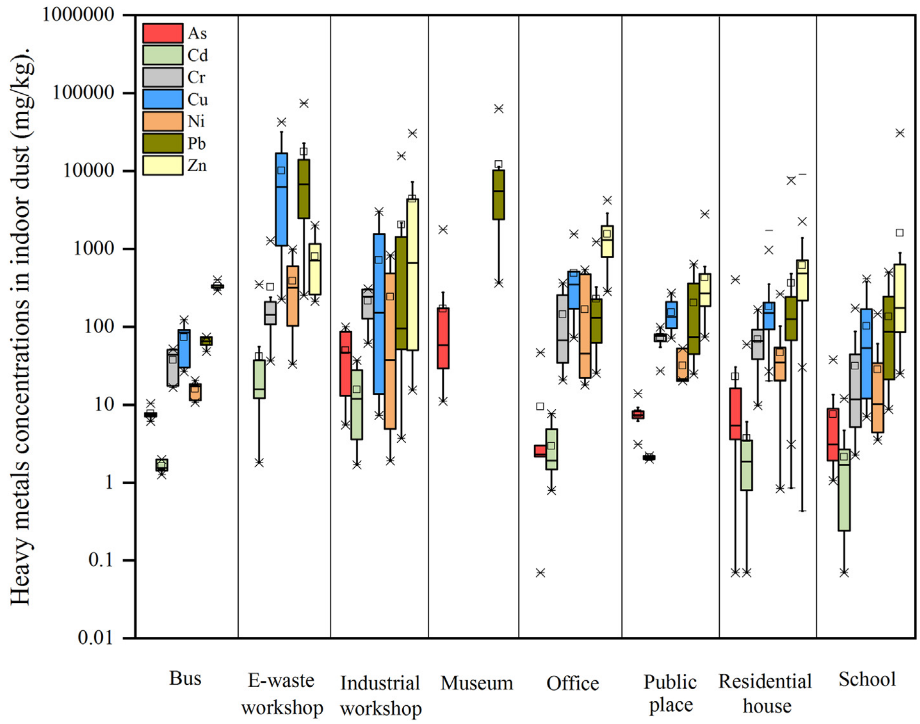

Fig. 4. Levels of heavy metals in indoor dust from different microenvironments.

bus ventilation system and street dust carried by passengers (Lei et al., 2016). Although the median concentration of Zn in settled bus dust exceeded the permissible standard, in the case of potential health risks, Pb ranked first. Therefore, greater attention should also be paid to the particular influence of Pb on commuters in the buses (Lei et al., 2016). Indoor dust in schools generally contained low concentrations of heavy metal contaminants, and none of the median concentrations exceeded the permissible levels.

3.4. Source identification

3.4.1. Possible sources of heavy metals in indoor dust

Indoor dust is a repository for environmental pollutants that may accumulate indoors from both external and internal sources over a long-term period (Yadav et al., 2019) (Fig. 5). Significant positive correlations were found between heavy metal concentrations in household and entryway dust ($p < 0.001$), thereby indicating that heavy metals in household dust may be derived from outdoor sources (Hassan, 2012). Soil is as a major contributor to house dust. The contribution of outdoor soils to indoor dust varies by heavy metal and is estimated to range from 20% to 45% (Trowbridge and Burmaster, 1997). Studies have also shown that 45–50% of house dust originates from soil and street dust (Fergusson et al., 1986). However, contrary to this view, there was no significant correlation in the contents of the heavy metals between indoor dust and soil, thereby indicating that soil might not be the only pollution source for heavy metals in indoor dust (Cao et al., 2016). Other possible outdoor sources are smelting, mining, other industrial activities (e.g., e-waste recycling and Pb-related industries), and automobile emissions. For example, industrial activities, traffic, and fossil fuel combustion are the primary sources that increase the heavy metal concentrations in school dust in Shiraz, southwest Iran (Moghtaderi et al., 2019). Trace elements such as Pb and Ni could be linked to vehicular emissions in Ogun State, Nigeria (Olujimi et al., 2015).

Dust generated within the house itself is also an important source of heavy metal exposure (Rasmussen et al., 2001). Indoor dust is a heterogeneous assortment of particles derived from exfoliated skin, hair, clothing, carpet fibers, paper, food, rubber, cosmetic products, plastics, building materials, smoking, cooking and heating (Turner and Ip, 2007). Rubber carpets and galvanized Fe roofs were identified as significant sources of Zn in Christchurch, New Zealand (Kim and Fergusson, 1993). The use of yellow pigments (Pb chromate) and other Pb pigments may significantly influence the Pb levels in house dust (Kim et al., 1998). The majority of heavy metals in indoor dust were mainly affected by the combined effects of exterior and interior sources. For example, building materials, paint, industrial activities and vehicle emissions were the factors attributed to the presence of heavy metals in classroom dust in Sri Serdang, Malaysia (Praveena et al., 2015).

3.4.2. Source identification methods

The sources of heavy metals in dust are generally identified by performing principal component analysis (PCA) and examining the enrichment factors (EFs), geochemical composition and isotopic composition.

PCA is widely used to reduce the original number of variables of contaminants and to analyze their sources (Harb et al., 2015; Paula et al., 2018; Sulaiman et al., 2017). Based on PCA, the relationships among heavy metals were found, thereby offering important information to identify the sources of house dust (Yoshinaga et al., 2014). PCA in combination with other statistical methods, such as multiple linear regression analysis, is commonly performed to obtain the mass apportionment of sources (Othman et al., 2019).

EFs are frequently applied to identify if soil and dust samples are contaminated by a certain element (Zhao et al., 2019). They can help to differentiate anthropogenic sources from natural sources (Liu et al., 2014). EF values >10 are attributed to anthropogenic origins, while EF values ≤10 indicate that the element is derived from natural sources (Biegalski et al., 1998). The comparison of EF values of heavy metals in indoor dust and outdoor dust can indicate the dominance of anthropogenic sources or natural sources (Rashed, 2008).

The geochemical composition of household dust is considered a possible fingerprint of a historic register of air quality (Torres-Sánchez et al., 2017). The organic composition of dust samples in terms of C, H, and N concentration ratios were defined to investigate the potential of

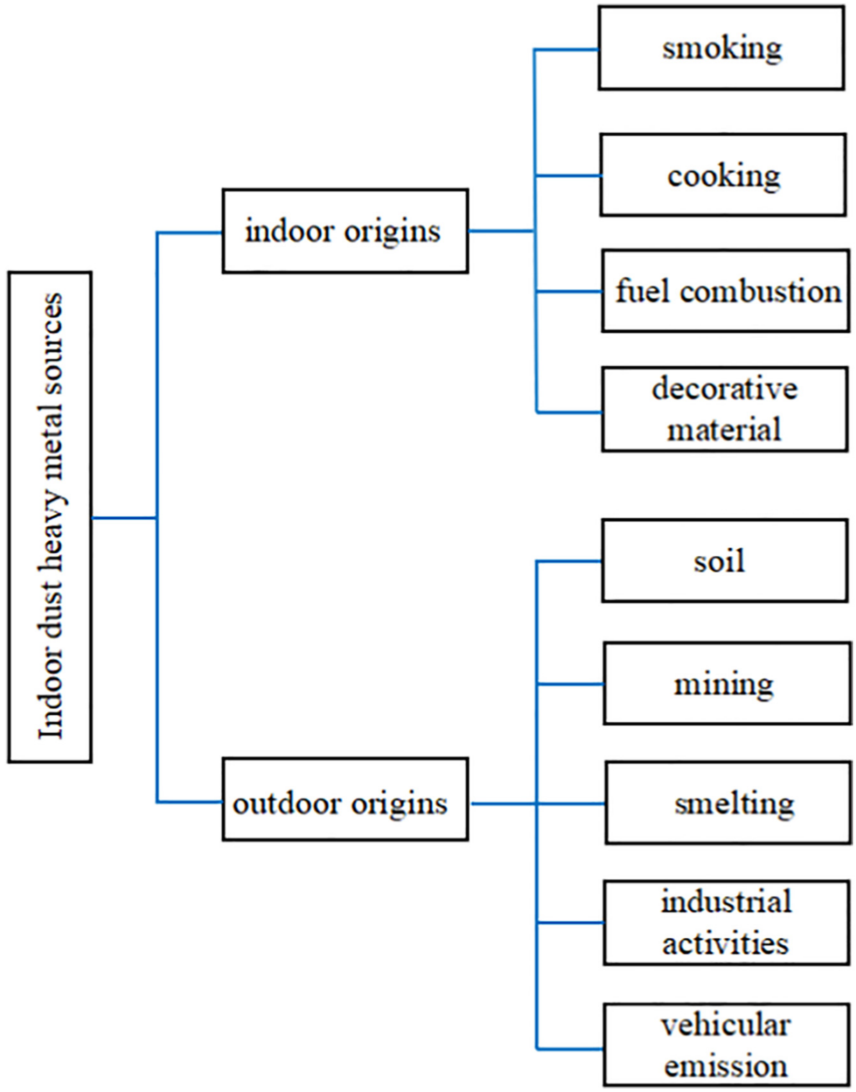

Fig. 5. Tree structure diagram of sources of heavy metals in indoor dust.

sourcing organic and inorganic components (Turner and Simmonds, 2006). The comparison of the chemical compositions of household dust and exterior dust and soils can further indicate the sources (Rasmussen et al., 2001). Furthermore, inorganic ions also provide important information on the origin of dust and are commonly employed in source identification and apportionment studies (Latif et al., 2009). Ca^{2+} and Mg^{2+} may indicate natural sources, K^+ is relevant to biomass combustion and meat cooking, and SO_4^{2-} and NO_3^- are related to combustion activities (Latif et al., 2009).

Isotopic composition methods have been used to distinguish the potential sources contributing to heavy metals in dust with adequate accuracy. For example, the contributions of Pb in house dust from exterior sources and interior sources are calculated by determining the isotopic ratios in street dust, house dust and paint used in the houses (Jabeen et al., 2001). Isotopic compositions of household dust can provide information on the sources, as well as the modes of formation, of Pb compounds (Miler and Gosar, 2019).

In addition, the possible sources of heavy metals in indoor dust in previous studies were seldom identified using positive matrix factorization (PMF). PMF is a receptor modeling tool developed in the early 1990s by Paatero and Tapper (1994). It is a convenient tool for quantifying source contributions. However, the PMF model is widely applied for soil (Guan et al., 2018), road dust (Guan et al., 2018) and $PM_{2.5}$ samples (Jain et al., 2017), but the application for indoor dust samples has been relatively limited. Thus, similar analytics should also be increasingly applied in indoor dust studies in the future.

To provide information about potential source of heavy metals in dust particles, their chemical speciation and atomic-level structure need to be determined. Advanced synchrotron radiation-based analytical techniques such as X-ray absorption fine structure (XAFS) spectroscopy, micro X-ray diffraction (μ-XRD), micro X-ray fluorescence (μ-XRF) techniques and microprobe techniques are applied to determine the speciation of heavy metals in dust particles (Beauchemin et al., 2011; Lu et al., 2014; MacLean et al., 2010; Rasmussen et al., 2008; Rasmussen et al., 2011; Zhou et al., 2019). The great advantage of these techniques is that they allow obtaining in situ molecular information without alteration of the original species distribution of heavy metals in particles.

3.5. In vitro bioaccessibility

The bioaccessibility of heavy metals is the actual fraction that can be dissolved partly or completely after contacting physiological fluids and is available for absorption, which may have toxic effects on the body later (Hu et al., 2018). Owing to the complex, time consuming, costly, and ethical issues of in vivo assays, in vitro models have generally been applied to evaluate the bioaccessible concentrations of heavy metals in dust in practice (Casteel et al., 2006; Hu et al., 2018; Liu et al., 2016; Tang et al., 2018). In vitro procedures simulate the sequential digestion of heavy metals in dust in simulated lung or gastrointestinal physiological fluids at a specific pH and temperature and for a specific duration (Bi et al., 2015).

Many heavy metals accumulate to higher concentrations in indoor dust and also have greater bioaccessibility in indoor dust compared with that in exterior soil (Argyraki, 2013). The main reason for this is that the elevated organic matter content in indoor dust increases the binding capability of heavy metals, thereby transforming inorganic compounds into more soluble metal-organic compounds (Rasmussen et al., 2008). The bioaccessible concentrations of some heavy metals in indoor dust were at high levels. The mean As bioaccessibility ranged from 75.4% to 83.2% in residential and school indoor dust in rural and urban areas of Hubei, China (Liu et al., 2016). The Pb bioaccessibility in Canadian house dust ranged from 63% to 81%, and high Pb bioaccessibility occurred in house dust with severe Pb contamination (Rasmussen et al., 2011). One possible reason for this is that a high proportion of bioaccessible compounds were used as pigments in older paints (Rasmussen et al., 2011). The average Pb bioaccessibility was 81.4% in house dust in an industrial town in China (Li et al., 2015). The Cd and Pb bioaccessibilities in indoor dust in the urban areas of Chengdu and Tianjin were >62.8% in the stomach phase (Li et al., 2016). The bioaccessibilities of Cu, Pb, and Zn in household dust in the United Kingdom were all approximately 80% (Turner and Simmonds, 2006).

The bioaccessibilities for heavy metals in household dust were greatly different. The mean bioaccessibilities of Cd, Zn, Pb, Cu, Ni, and Cr in household dust from urban areas in Chengdu, China were 80.20%, 70.00%, 43.90%, 39.70%, 30.50%, and 8.57%, respectively (Cheng et al., 2018). The median bioaccessibility ratios of heavy metals in house dust in Athens decreased in the order of Pb (78%) > Zn (73%) > Cd (46%) > Ni (42%) > Cu (37%) > Cr (27%) (Kelepertzis et al., 2019). The most likely explanation for this is that different forms of these heavy metals may occur in the digestion system (Wang et al., 2016).

There are differences in the heavy metal bioaccessibilities between the stomach and intestinal phases. The bioaccessibilities of some heavy metals significantly decreased from the stomach to the intestine (Huang et al., 2014b). Pb bioaccessibility in the stomach phase ranged from 17.6% to 76.1%, while it varied from 1.2% to 21.8% in the more alkaline intestinal phase (Bi et al., 2015). The bioaccessibilities of Cd, Ni, Pb, and Zn in the stomach were higher than those in the intestine (Turner and Ip, 2007). Because of the alkaline and carbonate-rich environment in the intestine, insoluble species precipitate and cations reabsorb to preexisting or altered sites on the solid surface (Turner and Ip, 2007).

In vitro test systems are simpler, more reproducible, time-saving, and money-saving than in vivo animal models, but also bring uncertainties. The methods used to extract and digest total heavy metals vary among different in vitro bioaccessibility procedures (Li et al., 2014a, 2014b). Thus, the in vitro bioaccessibility of heavy metals is impacted by the procedure (Hu et al., 2018). Furthermore, the in vitro bioaccessibility of heavy metals shows metal speciation, particle size, pollution source, and sample matrix dependence (Bi et al., 2015). Further studies to identify the variables controlling heavy metal bioaccessibility in dust are still needed. In vitro bioaccessibility procedures should be further optimized and validated for risk-based assessments.

3.6. Factors influencing heavy metal accumulation in indoor dust

3.6.1. Seasonal variation

Distinct seasons are different in aspects such as weather stability, wind velocity, and wind direction. Seasonal variation is a critical factor affecting house dust concentrations of heavy metals. Habil et al. (2013) found that the highest enrichment of Cd in classroom dust was in summer, followed by the monsoon season and winter in a preschool located in Agra City, India (Habil et al., 2013). However, other researchers have found that indoor heavy metal concentrations are relatively high in seasons with a temperate climate. El-Desoky et al. (2014) found that Pb in indoor dust exhibited decreasing concentrations in the order of spring > summer > winter > autumn in Riyadh, Saudi Arabia. The mean values of Cd, Cr, Cu, Ni, Pb and Zn in temperate months were significantly higher than those in other months ($p < 0.05$) in Ahvaz, Iran (Neisi et al., 2016). The average seasonal concentration of Pb in indoor dust was in the decreasing order of autumn > winter > spring > summer in Kerman City, southeast Iran (Abbasnejad and Abbasnejad, 2019). Thus, the influence of seasonal variation on heavy metal levels in indoor dust varies with the geographical location, and there are no clear conclusions at present.

3.6.2. Dust particle size

The particle size of indoor dust varies widely, ranging from tens of microns to <1 mm, and some smaller particles attach to larger particles (Zhou et al., 2019). The <150 mm particle size fraction of dust was selected to minimize the contribution of exterior soil to the indoor dust composition (Paula et al., 2018). The size dependence of the heavy metal concentrations is of special interest because the specific surface area of particles is indirectly proportional to the particle size. Furthermore, the total C and mineral components are not uniformly distributed in the various size fractions of house dust (Lanzerstorfer, 2017), which play a role in heavy metal partitioning among mineral and organic phases and in the digestion process. Studies have reported that the concentrations of heavy metals increase significantly with a decrease in particle size (Niu et al., 2010). The maximum concentrations of heavy metals were found in the size fraction of 200 μm. The concentrations of heavy metals vary with particle size. For example, Pb, Ni, Cd, Zn, Co, and Cr were found in significantly ($p < 0.01$) higher concentrations in dust with small particle sizes (< 38 μm), whereas Al, Fe, and Cu were detected in significantly ($p < 0.01$) higher concentrations in dust with large particle sizes (> 45–63 μm) (Hassan, 2012).

Particle size is a critical factor in the risk assessment of human exposure to heavy metals in indoor dust (Cao et al., 2012). Smaller particles have lower deposition velocities than coarse particles, and thus can remain in the air for a longer amount of time, thereby potentially impacting human respiration (Gustafsson et al., 2018). When estimating the bioaccessibility of heavy metals in indoor dust, fine fractions (< 63 μm) are usually taken into account (Turner and Ip, 2007). Particles smaller than 10 μm are particularly hazardous owing to their minute sizes, which facilitate entry into the lungs (Huang et al., 2010). Industrial dust particles, such as solvent-based paint dust particles, are very fine in size (< 1 μm) and have high Pb and Zn concentrations (Huang et al., 2010). To fully understand the potential health effects of inhalation exposure to household dust, it is crucial to achieve a more complete characterization of the very fine fractions (Lanzerstorfer, 2017), especially the respirable dust fraction (Gustafsson et al., 2018). Studying heavy metal concentrations in dust samples with a unified particle size can provide results that can be more realistically compared with samples from different environments (Al-Rajhl and Seaward, 1996).

3.6.3. Geographical location

Indoor dust heavy metal concentrations obtained from different regions were compared to verify the regional differences in household contamination, particularly in areas with disturbance of anthropogenic activities, as well as to identify the controlling factors.

There are regional differences between urban and rural areas in geology and human activities, such as industrial emissions, traffic emissions, and cultural practices, which may result in variation in the heavy metal content in indoor dust. For example, Iwegbue et al. (2017) analyzed the spatiotemporal distribution of metals in household dust from rural, semi-urban, and urban environments in the Niger Delta, Nigeria. Similarly, Jelenska et al. (2017) characterized the magnetic, chemical and microscopic characteristics of dust from the outer city center, out of the center, and the very center of the city in Warsaw, Poland. Concentrations of Pb, Cu, Zn and Fe in dust samples in urban schools were between two to four times higher than those in semi-rural schools (Akinwunmi et al., 2017). Only Ni showed significant differences in concentrations in dust from urban, suburban, and rural areas in Istanbul, Turkey (Kurt-Karakus, 2012).

Comparison of indoor dust pollution in contaminated areas with that in uncontaminated areas has been conducted to identify industrial and traffic pollution. Sampling sites in urban, residential, and residential areas near industrial areas were selected to perform a comparative study (Hassan, 2012). Industrial, heavy traffic, and residential zones in Ahvaz were investigated (Neisi et al., 2016). *E*-waste recycling, urban, and rural areas were selected to illustrate the spatial characteristics and to further evaluate human exposure risks (He et al., 2017). Roadside and residentially located schools were identified to assess the extent of contamination and sources of heavy metals in classroom dust (Habil et al., 2013).

In addition, dust samples were collected to investigate the variation in terms of land use and function, including commercial districts, industrial districts, traffic pivotal districts, residential districts, educational districts, scenic parks and peri-urban districts (Huang et al., 2014b; Zhou et al., 2019).

3.6.4. Heavily polluting industries

Most investigations on heavy metal contamination in household dust are centered on buildings located close to pollution sources. Significant contributions from various industrial sources is evident, e.g., electric and electronic waste (e-waste) recycling, mining, smelters, industrial activities and traffic.

3.6.4.1. E-waste recycling. Electric and electronic waste, or e-waste, represents an emerging environmental problem. Developing countries such as China and India face a rapidly increasing amount of e-waste that is imported illegally from developed countries for recycling (Zeng et al., 2017). E-waste recycling activities are regarded as significant sources of contaminants in workshops and homes in e-waste recycling areas. The concentrations of Cd, Cr and Pb decreased with increasing distance from e-waste recycling centers (Wu et al., 2016). The levels of Cd, Cu, Ni, Pb and Zn in dust decreased after relocating e-waste sites (Yu et al., 2019). The levels of As, Cd and Pb in indoor dust in an e-waste recycling area in Taizhou, which is one of the two largest primitive e-waste recycling centers in eastern China, exceeded the corresponding background values by nearly 36, 13, and 6 times, respectively (Wu et al., 2016). The highest concentrations of Pb and Cu were 22,900 mg/kg and 42,700 mg/kg, respectively, in television dismantling workshop dust (Deng et al., 2014).

3.6.4.2. Mining and smelting. Yang et al. (2015) reported that As and Pb in house dust surrounding a phosphate mine were enriched. The contamination associated with mining and metallurgical activities was increased by the proximity to the mines (Fontúrbel et al., 2011). Almost all the studied biomarkers (blood, urine, hair and nails) of residents were significantly altered compared with those of the control population in the mining area of Panasqueira, Portugal (Paula et al., 2018). Mining activities adversely affect the health of miners and the communities living near mine sites, and these effects may persist even when the mine is abandoned.

Heavy metals emitted from the smelting process and transported by the atmosphere are the main source of pollution in indoor dust. The highest Pb concentration (314.1 mg/kg) in dust has been found in samples collected in smelters ovens and mechanical sites in Karak Industrial Estate, Jordan (Al-Khashman, 2004). Extremely elevated concentrations of Cd (2.2–124.0 mg/kg), Pb (220.0–6348.0 mg/kg) and Zn (256.0–8245.0 mg/kg) were observed in a former Zn smelting area in Guizhou, China (Bi et al., 2015). The degree of heavy metal pollution decreased gradually with increasing distance to the smelter, with the highest levels corresponding to the prevailing wind direction (Soto-Jiménez and Flegal, 2011; Xie et al., 2013). As, Cd and Pb levels in indoor dust were found to be much higher in Hettstedt, which is a city in eastern Germany with an over 800 y history of mining and smelting of non-ferrous metals, even though the Pb smelter had been closed for over 10 y (Meyer et al., 1999b). Thus, it is essential to maintain a healthy distance between metallurgical areas and urban areas to mitigate the deleterious impact and health effects associated with smelting.

3.6.4.3. Other industrial activities. Previous studies have demonstrated that the surroundings of industrial areas are the areas that have the highest heavy metal concentrations. Industrial processes invariably generate fugitive dust during operation. The levels of Cu, Pb and Zn were elevated in house dust near industrialized areas in Taejon, Korea (Kim et al., 1998). Dust from a paint manufacturing plant had high Cu (mean 1550 mg/kg), Pb (mean 15,680 mg/kg), and Zn (mean 30,460 mg/kg) concentrations (Huang et al., 2010). One possible reason for the high Zn concentration in paint dust is that Zn is normally used as a metal pigment for paint coating and anti-corrosive primers (Muller, 2001). The surgical instrument manufacturing industry has different sub-sections that produce coarse and fine dust contaminated with heavy metals, especially Cr and Ni (Junaid et al., 2016).

3.6.5. Traffic load and special areas

The road grade around a house is an important factor influencing the heavy metal concentrations in indoor dust (Lin et al., 2015). The burdens of Cd, Cu, Pb and Zn in areas with heavy traffic are significantly higher than those in other districts in Hong Kong, China (Tong and Lam, 2000). The heavy traffic density and usage of leaded gasoline are external sources of Pb deposition in indoor dust. The concentrations of Pb and Zn decrease as the distance of the houses from major roads increases in urban areas (Latif et al., 2009).

In order to preserve natural history collections, many different and often toxic products have been used, which are likely to pollute the indoor air and settled dust in museums (Marcotte et al., 2014). Settled dust in the Natural History Museum of Rouen, France was significantly contaminated with As, Hg and Pb (Marcotte et al., 2014). The highest gaseous Hg concentrations were found in the herbarium storage area, especially when manipulating herbarium sheets accompanied by mercuric chloride (Marcotte, 2017).

3.6.6. House characteristics

In addition to outdoor sources, domestic factors such as house age, floor level, floor cover, cigarette smoking, ventilation, energy usage and cleanliness could explain the differences in house dust originating from indoors.

3.6.6.1. House age. House age only affects the accumulation of some heavy metals in indoor dust. According to Rasmussen et al. (2013), the relationships between house age and dust heavy metal concentrations were significant for Pb, Cd and Zn ($p < 0.001$), but not for As, Cr, Cu and Ni. Higher Pb concentrations occurred in dust samples of older homes in Ottawa, Canada (Rasmussen et al., 2001). Similarly, the Pb levels in indoor dust in older houses were significantly higher than those in newer houses (Kim et al., 1998). Kelepertzis et al. (2019) observed that only Pb displayed a trend of increasing concentration with house age. It was estimated that approximately 45% of the Pb was derived from paint in old houses (Fergusson and Schroeder, 1985). The concentrations of Cr, Cd, Cu, Ni, Pb and Zn displayed an increased trend with the time of last painting and house age (Cheng et al., 2018). Thus, we can infer that the deterioration and peeling of paints on the walls of old buildings settles as indoor dust, thereby causing high Pb concentrations in indoor dust (Tan et al., 2016). Blood lead levels (BLLs) had a significant positive association with the age of housing, with children living in households constructed prior to 1945 being more likely to have higher BLLs (Safruk et al., 2017). It has also been found that houses younger than 20 y constructed of concrete may not be considered old enough to highly impact the heavy metal concentrations in house dust (Shraim et al., 2016).

3.6.6.2. Floor level and floor cover. There is a negative correlation between heavy metal (Cr, Cd, Cu, Ni, Pb and Zn) concentrations and floor level (Cheng et al., 2018). The mean concentrations of As, Cu, Ni, Pb and Zn in the indoor dust of bungalows were found to be significantly higher than those in storied houses (Lin et al., 2015). The concentrations of Pb in indoor dust in the first floor tended to exceed the concentrations in other floors (Rashed, 2008). This difference might be attributed to the proximity of the first floor to the street and traffic emissions (Rashed, 2008). The concentrations of Cd, Cr, Ni, Pb and Zn in household dust were affected by the lower residential layers (Cheng et al., 2018). The concentrations of Cd, Cr, Ni, Pb and Zn on the 7th floor were significantly lower than those on the other floors of a seven floor building (Zhou et al., 2019). It is possible that higher floors lead to changes in the heavy metal pollution patterns of indoor dust (Zhou et al., 2019). However, some studies found that heavy metals in indoor dust showed a regular increase with vertical distance (Rashed, 2008). The concentrations of As, Cd, Cr, Ni, Pb and Zn at different floors of the building demonstrated a significant increase from floor 2 toward floor 13 in Hefei, China (Ali et al., 2019). One possible reason for this is that most of the particles were found to be <100 μm, and decreased significantly with the increase in floor altitude (Ali et al., 2019). Thus, further research is needed to conclude the impact of floor level on indoor dust heavy metal concentrations.

Carpets provide a reservoir where heavy metals that have an affinity for organic-rich particles may become more concentrated as the dust ages (Rasmussen et al., 2018). The concentrations of As, Cd, Cr, Cu, Ni and Zn were approximately 1.4–2.1 times higher in the dust from carpeted homes than in dust from non-carpeted homes (Rasmussen et al., 2018). Apartments floored with plastic carpets had higher concentrations of Cd, Cr, Cu, Ni, Pb and Zn than those with ceramic tiles (Iwegbue et al., 2017).

3.6.6.3. Cigarette smoking. Opinions remain divided about the influence of smoking on the amount of indoor dust pollutants. According to Latif et al. (2009), dust from households with smokers have high concentrations of Cd and Ni (Latif et al., 2009). Similarly, smoking is an important factor of heavy metal (Pb, Zn and Cd in particular) enrichment in household dust (Cheng et al., 2018). However, comparisons of non-smokers' and smokers' homes showed no difference ($0.15 \le p \le 0.97$) in dust heavy metals (As, Cd, Cr, Cu, Ni, Pb and Zn) concentrations in Canada (Rasmussen et al., 2013). The explanation for this is that associations between heavy metal loadings and smoking activity are mainly driven by increased dust loading in homes occupied by smokers (Rasmussen et al., 2013). Furthermore, smoking may need to be limited or conducted under more suitable conditions (such as with good ventilation) in indoor environments.

3.6.6.4. Ventilation. Ventilation through open windows was found to be a possible factor that contributes to heavy metal accumulation on windows, floors and fans (Praveena et al., 2015). Homes that did not have their windows open often had a lower level of contaminants in house dust (Tong and Lam, 2000). Wind-blown dust from surface soil and road dust is probably the main contributor of heavy metals in indoor dust. However, increased levels of As, Cd and Pb were found in damp households compared with those in dry households (Meyer et al., 1999b). It may be possible that a higher indoor humidity promotes particle coagulation and condensation, and thus increases the deposition velocities of metal-containing particles (Meyer et al., 1999b). Higher Hg concentrations may accumulate in the indoor dust of damp, poorly ventilated homes (Rasmussen et al., 2001). Inadequate exchange between indoor air and outdoor air can result in an increased indoor fungal concentration (Garrett et al., 1998). Fungi and other lower plants are capable of accumulating high concentrations of Hg (Lodenius and Herranen, 1981).

3.6.6.5. Energy use and cleanliness. The style of heating was also found to influence the levels of certain heavy metals in household dust. Dust of electrically heated houses tended to have higher Pb and Hg concentrations than dust of gas-heated or oil-heated houses (Rasmussen et al., 2001). This is possibly due to the associated differences in air circulation and use of particle filters (Rasmussen et al., 2001). Hg concentrations in indoor dust were highly significantly associated with coal combustion (Lin et al., 2017). Coal combustion is the largest source of Hg in the environment (Zhao and Luo, 2017). Pollutants containing Hg are discharged into the environment from thermal power facilities, which are the heating equipment of burning coal (Klein et al., 1975). The use of metal furnaces fueled by diesel installed in living spaces for heating significantly increased the concentrations of Ni, Pb and Zn in house dust (Al-Momani, 2007).

Cleaning frequency is also expected to be a significant factor affecting the heavy metal accumulation in indoor dust (Praveena et al., 2015). There were negative correlations between the heavy metal (Cd, Cr, Ni, Pb and Zn) concentrations and sweeping frequency (Cheng et al., 2018). Occupants who sweep their floors or furniture on a daily basis or use vacuum cleaners had lower levels of heavy metals inside their houses (Tong and Lam, 2000). Coarser fractions of dust are preferentially removed during cleaning. Thus, maintaining interior sanitation would be a more effective measure to reduce the detrimental health effects.

3.7. Health risk assessment

Dust plays an important role in human health owing to the complex chemistry and the possibility of re-emission. Heavy metals in indoor dust are important predictor variables for heavy metal levels in biomarkers (Junaid et al., 2016; Lucas et al., 2015; Reis et al., 2018; Thornton et al., 1990). Cr in industrial indoor dust and urine/saliva samples of exposed workers in surgical instrument manufacturing industry showed significant positive correlations (Junaid et al., 2016). Zn concentrations in indoor dust were good predictor variables for toenail Zn contents (Reis et al., 2018). Dust Pb in home environment was an important predictor of blood lead concentrations in young children (Thornton et al., 1990). The potential human health risks from heavy metals associated with dust are generally assessed based on the US EPA health risk model (Cao et al., 2016; Cheng et al., 2018; Neisi et al., 2016; Praveena et al., 2015).

3.7.1. Exposure assessments and risk levels

The potential human health risks from heavy metals associated with indoor dust at different locations vary greatly. The non-carcinogenic risks (hazard quotient; HQ) and carcinogenic risks (cancer risk; CR) for children in urban areas were 1.59–1.95 times greater than those for children in rural areas of Hubei, China (Liu et al., 2016). Residents in industrial areas had higher potential health risks than those in urban or rural areas (Eqani et al., 2016a; Tan et al., 2016; Zheng et al., 2013). There are also significant differences in the health risks associated with heavy metals in dust of different ages (Lei et al., 2016). Children had higher risks than adults of exposure to heavy metals via indoor dust intake owing to their higher frequency of hand-to-mouth activities (Li et al., 2016). The HQ and CR values for children 3 y to 5 y of age were 1.40–1.47 times greater than those for children 6 y to 9 y of age in urban areas of Hubei, China (Liu et al., 2016).

Heavy metal exposure via dust may pose significant health risks to the population in e-waste recycling regions (Wu et al., 2016). The HQ values of Pb were 2.27 and 9.52 for adults and toddlers, respectively, in e-waste workshops (Xu et al., 2015). The HQ value (1.56) for children's exposure to Pb in dust from electronic workshops suggested a considerable non-carcinogenic risk (Iwegbue et al., 2018). The potential risks of heavy metal exposure from dust ingestion for young children from an e-waste recycling area were remarkably high, with HQ values of Pb of 11.10, Cu of 2.78, and Cd of 1.16 (Zheng et al., 2013). Thus, Pb was the most notable pollutant that posed a significant risk in the e-waste recycling area, which warrants further attention (He et al., 2017). There could be significant health risk implications in some industrial sectors. The CRs of Cr and Cd were 3.0×10^{-1} and 4.1×10^{-2} in exposed workers from surgical instrument manufacturing industries, respectively, which were much higher than the safety limits ($1 \times 10^{-6} < CR < 1 \times 10^{-4}$) reported by the US EPA (Junaid et al., 2016).

Dust exposure in some underdeveloped nations poses several health risks to large populations because the issues of severe environmental degradation, poor hygiene, and dust exposure have been neglected (Mohmand et al., 2015). For example, exposure to Pb in house dust has potential non-carcinogenic risks in children and adults in mountainous areas of Pakistan (Eqani et al., 2016b). Cr and Pb are the main contaminants that pose a non-carcinogenic risk to the human population, while Cr, Cd, Ni, and Pb pose higher carcinogenic risks to the human population in Nepal (Yadav et al., 2019).

For children's health risk assessment in most of the schools, the non-carcinogenic or the carcinogenic risks derived from indoor dust were within the safe range (Lu et al., 2014; Moghtaderi et al., 2019; Othman et al., 2019). However, long-term persistence of leaded gasoline, building materials, interior paint, and location near major roads may also be causes of health risks of Pb in school. The maximum HQ for the non-carcinogenic risk of Pb in classroom dust for children was greater than the safe value of 1 in Sri Serdang, Malaysia (Praveena et al., 2015).

Most studies identified oral ingestion as the most critical indoor dust exposure route for humans, followed by dermal uptake and inhalation (Kurt-Karakus, 2012; Olujimi et al., 2015; Zheng et al., 2013), and it is especially true for children as they frequently exhibit hand-to-mouth behavior. Heavy metal intake via dust inhalation was <1% for both Chengdu and Tianjin residents (Li et al., 2016). Some research has even suggested that compared to ingestion and dermal contact exposure, exposure through inhalation is almost negligible (Kurt-Karakus, 2012). Studies have also shown that there were great variations in the risks from three exposure pathways for different heavy metals. Ingestion is an important pathway of exposure to Pb, Zn, Cu and Ni in household dust for residents, and dermal contact was identified as a main route for Cr and Cd exposure in urban area in Chengdu, China (Cheng et al., 2018). The inhalation of Hg vapor in dust is the main route of Hg exposure for adults and children (Lin et al., 2017).

3.7.2. Uncertainty analysis

The potential health risks may be overestimated by assessments based on total heavy metal concentrations in indoor dust. The health risk taking bioaccessibility into account was only 50.8–59.8% of that obtained without consideration of bioaccessibility (Liu et al., 2016). The bioavailability/bioaccessibility of heavy metals is regarded to be more reliable and accurate for human risk assessment. On the basis of evaluating the potential health risks by pollutant investigation, further studies on metal biomarkers in body fluids such as blood and urine are necessary to verify the adverse health effects of heavy metals (Neisi et al., 2016). Moreover, health risk estimation is usually accomplished using standardized parameters that are obtained from the US EPA. Accordingly, future research will be required to obtain local parameters (e.g., body weight and exposure frequency) based on the study area or country in order to obtain an accurate health risk assessment. The precise health risk exposure is a function of numerous media, including air, soil, food, and drinking water. Thus, assessing health risks through the single pathway of dust is not sufficient; cumulative and aggregative exposure of heavy metals via multiple pathways might be more reliable.

3.8. Prediction of blood lead levels

The World Health Organization estimates that the sources of Pb exposure in children include dust and soil (45%), food (47%), water (6%), and air (1%) (Han et al., 2018). The Pb concentration in household dust has been found to be strongly associated with the geometric mean of children's BLLs (Safruk et al., 2017); however, this relationship occurs more remarkably and frequently in communities with significant Pb emissions from point sources. The US EPA's integrated exposure uptake biokinetic (IEUBK) model is an empirically validated model (White et al., 1998). It has been commonly used to predict BLLs from Pb exposure through multiple environmental media in children aged 0–7 y old. The parameters of exposure model for Pb concentrations in multiple environmental media, daily consumption rates (e.g., inhalation rates, and ingestion rates of soil/dust, grain, vegetables, and water), and absorption fractions from different environmental media can be found in Table S2 and S3. There was an assumption that exposure pathways including soil, water, and diet in different regions would be the same, with only the effect of dust taken into account. The predicted mean BLLs for children aged 1–2 y (the most susceptible populations based on the prediction results) in various regions are presented in Fig. 6.

49.8%, 36.8% and 14.4% of the study sites showed BLLs of children in the age group of 1–2 years exceeding 35 μg/L (threshold limit in Germany), 50 μg/L (threshold limit in the USA), or 100 μg/L (threshold limit in China), respectively. The BLLs far exceeded 100 μg/L in some regions with significant point sources, such as a Cu mine in Zambia (Ndilila et al., 2014); e-waste maintenance shops in Western Oromia, Ethiopia (Getachew et al., 2019); e-waste workshops in Wenling, China (Xu et al., 2015); a paint manufacturing plant in Shah Alam, Selangor, Malaysia (Huang et al., 2010); and a smelter in Yunnan, China (Xie et al., 2013). Pb exposure still represents a major contributor to children's blood Pb poisoning in many low-income and middle-income countries. Children's BLLs in China, India, Iran, Pakistan, Zambia, etc. were relatively high (Fig. 6). Previous studies indicated that intensive anthropogenic activities, mainly Pb-acid battery manufacturing, Pb mining and smelting, and other industrial activities, are responsible for relatively higher levels of Pb exposure in eastern areas (Li et al., 2014a, 2014b) based on the geographic division of Zhang et al. (2001). It has been reported that even if the BLL is lower than 100 μg/L or even 30 μg/L, significant impacts on height, breast, and pubic hair development during puberty could occur (Wilhelm et al., 2010). To date, there is no internationally defined blood Pb poisoning effect threshold. It is necessary to implement intervention policies to ensure that no children live or spend significant time in homes, buildings or other environments with lead-exposure hazards.

4. Conclusion and perspectives

Through a systematic literature review of English databases, this study had identified 179 research articles with useful data and provides a description of the overall pollution situation of heavy metals in indoor dust at a global scale. Based on the WHO standard for soil quality, Cu and Zn pollutions in indoor dust were the heaviest and should receive prioritized remediation attention. There was clear spatial variation in the heavy metal concentrations at a global scale. Scattered abnormal peaks demonstrated that mining, smelting, e-waste recycling, and industrial production were critical factors affecting heavy metal accumulation in indoor dust. The bioaccessible concentrations of some heavy metals in indoor dust were at high levels, such as Pb, Cd, Cu and Zn. Heavy metals in the indoor environment arise from a number of contributing and polluting sources, and are greatly influenced by many factors, e.g. seasonal variation, geographical location, industrial discharges, and house characteristics. There is a need to quantitatively identify the sources and influencing factors of heavy metal accumulation in indoor dust in future studies.

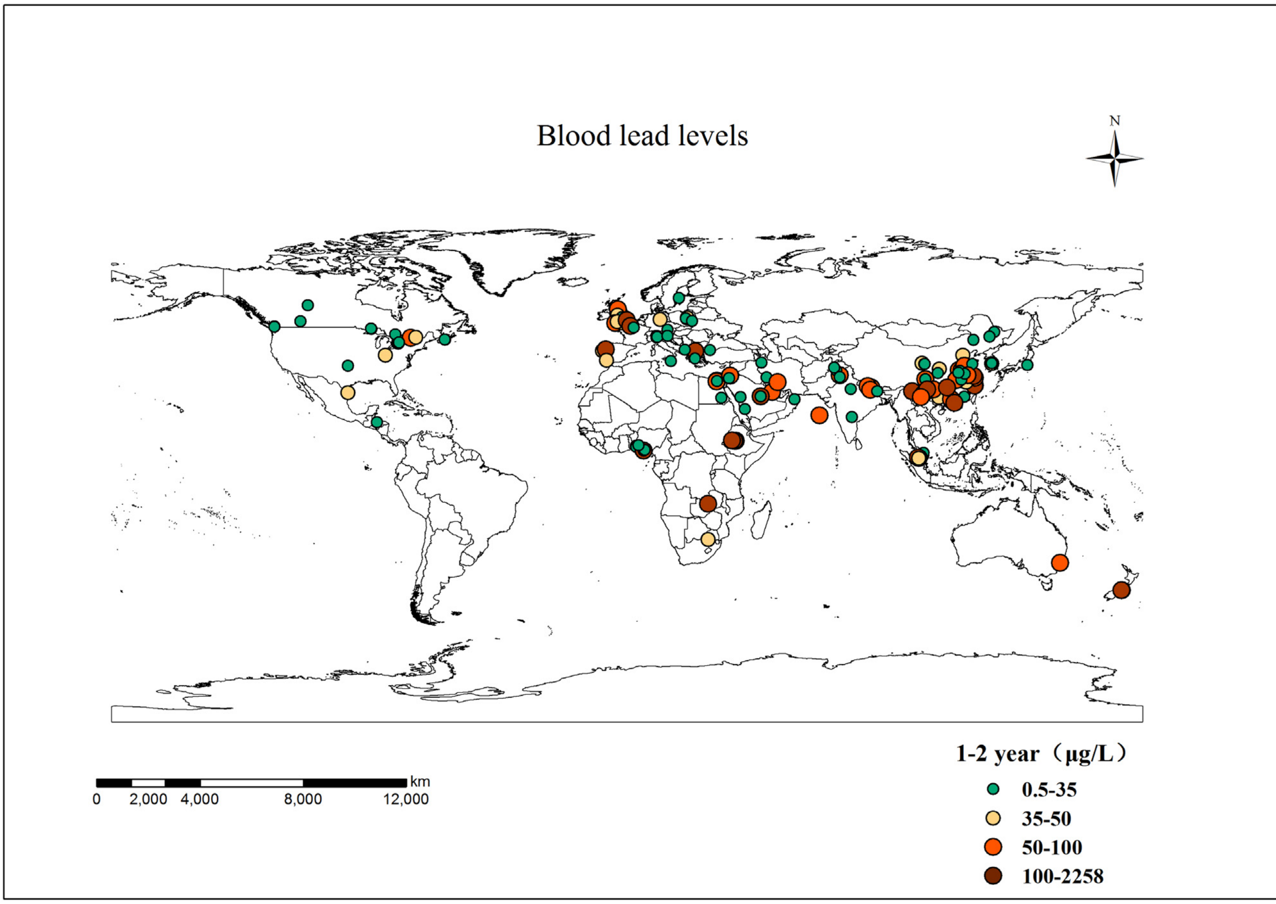

Fig. 6. Spatial distribution of the predicted children's blood lead levels in 1–2 y old children.

According to the health risk assessment, Pb had the highest risk in e-waste recycling areas. Based on the predicted BLLs, Pb exposure still represents a major contributor to children's blood Pb poisoning in many developing countries, particularly in China. At present, there is no statutory limit for heavy metals in dust or hygiene standards to limit the intake of contaminated indoor dust. A significant reduction in childhood exposure to heavy metals will not be accomplished only through lowering outdoor soil cleanup criteria and guidelines, but also through increased attention to indoor sources of exposure. It is suggested that indoor dust monitoring should be conducted regularly, and dust heavy metal hazard standards are needed to reduce human exposure.

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CRediT authorship contribution statement

Taoran Shi: Writing - original draft, Funding acquisition
Yuheng Wang: Supervision, Language editing

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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