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Review

To breathe or not to breathe: Inhalational exposure to heavy metals and related health risk

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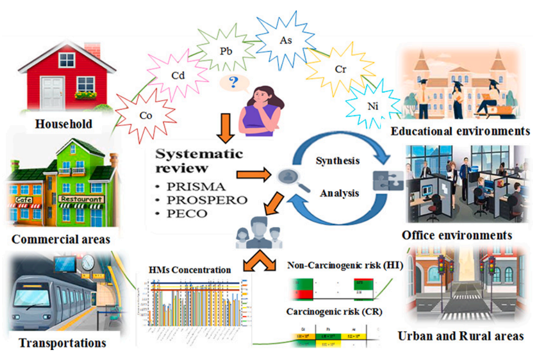
HIGHLIGHTS

- A systematic review was undertaken on environmental exposure to heavy metals (HMs).
- HM concentrations in most households were higher than the WHO guideline values.
- The HQ values of HMs in household and commercial environments were significant.
- The CR values in all the environments investigated exceeded the acceptable level.
- A definite carcinogenic risk was stated for Cr in most of investigated environments.

GRAPHICAL ABSTRACT



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ABSTRACT

This study reviewed scientific literature on inhalation exposure to heavy metals (HMs) in various indoor and outdoor environments and related carcinogenic and non-carcinogenic risk. A systematic search in Web of Science, Scopus, PubMed, Embase, and Medline databases yielded 712 results and 43 articles met the requirements of the Population, Exposure, Comparator, and Outcomes (PECO) criteria. Results revealed that HM concentrations in most households exceeded the World Health Organization (WHO) guideline values, indicating moderate pollution and dominant anthropogenic emission sources of HMs. In the analyzed schools, universities, and offices low to moderate levels of air pollution with HMs were revealed, while in commercial environments high levels of air pollution were stated. The non-carcinogenic risk due to inhalation HM exposure exceeded the acceptable level of 1 in households, cafes, hospitals, restaurants, and metros. The carcinogenic risk for As and Cr in households, for Cd, Cr, Ni, As, and Co in educational environments, for Pb, Cd, Cr, and Co in offices and commercial environments, and for Ni in metros exceeded the acceptable level of 1×10^{-4}. Carcinogenic risk was revealed to be

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higher indoors than outdoors. This review advocates for fast and effective actions to reduce HM exposure for safer breathing.

1. Introduction

The term heavy metals (HMs) refers to metals and metalloids with densities of >4000 kg m^{-3} (Vardhan et al., 2019). They are not biodegradable, persistent in the environment, hazardous in small amounts, and bioaccumulative in tissues. Consequently, they pose a substantial risk to both ecological systems and human health (Abdulaziz and Musayev, 2017). The adverse effects of exposure to HMs on human health are well documented and cover a wide range of health issues, including kidney and liver damage, respiratory illnesses, neurological disorders, anemia, skin lesions, kidney diseases, congenital deformity, and cancer (Sall et al., 2020). Metals exist in the Earth's crust in background concentrations, but most enrich the environment with their concentrations through anthropogenic processes, including metal refining and mining, and industrial processes such as electroplating, leather tanning, metalworking, production of electronics, petrochemicals, and chemicals (Fig. 1). Furthermore, the burning of fossil fuels, vehicle emissions, cement manufacturing, and application of metal-containing chemicals in agriculture may discharge HMs into the atmosphere (Rehman et al., 2021). Additionally, the geochemical cycles of HMs can be substantially altered by the open disposal of industrial effluents and waste, resulting in their discharge into the environment (Ali and Khan, 2018).

Ambient air is a critical medium to assess the environmental condition and status of HM contamination due to its ability to facilitate long-distance transport of HM-contaminated particulate matter (PM) (Bilos et al., 2001). Considering the outdoor environment is a crucial aspect that influences low indoor air quality (IAQ), so indoor air pollutants may originate from different sources and have varying concentrations compared to the outdoor environment (Fromme et al., 2008). A multitude of studies compare the concentrations of particulate matter (PM) in indoor with outdoor air in households (Massey et al., 2009; Pavilonis et al., 2013), schools (Guo et al., 2008; Madureira et al., 2012), museums (Cao et al., 2011), and offices (Quang et al., 2013). Research on the link between indoor and outdoor pollution can show how effectively the building envelope protects inhabitants from external PM and how the ventilation system's efficacy in dispersing indoor-generated particles (Guo et al., 2008).

The US Environmental Protection Agency (USEPA) developed the Human Health Risk Assessment (HHRA) methodology to calculate the human health risk associated with exposure to contaminants. Indoor air quality has become increasingly important due to the fact that most people spend around 90 % of their time in indoor settings, such as residences, educational institutions, and workplaces (Klepeis et al., 2001; Tran et al., 2012). According to Rashed (2008) and Turner (2011), fine (≤100 μm) deposited airborne particles in indoor environments constitute indoor dust. Additionally, indoor dust can contain contaminants from both internal and external sources. Research has indicated that indoor dust can transport both inorganic and organic pollutants, including HMs, pesticides, polychlorobiphenyls, and polycyclic aromatic hydrocarbons (Tong and Lam, 2000; Wang et al., 2015).

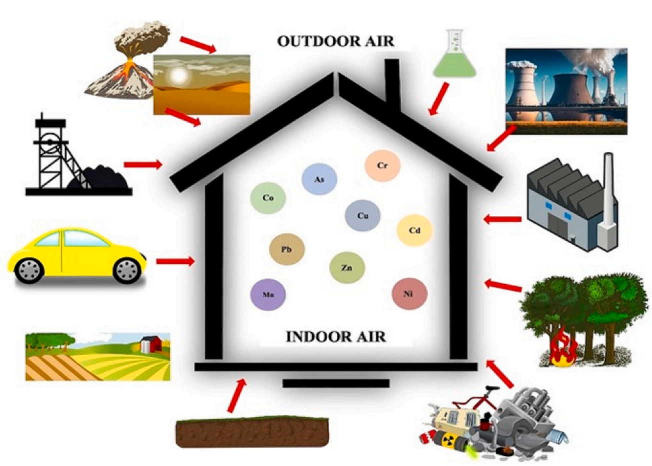

Fig. 1. Sources of exposure to HMs due to the outdoor air pollution.

Because of their high toxicity, lack of biodegradability, and detrimental effects on human health, HMs and other pollutants found in indoor dust warrant additional investigation (Darus et al., 2012; Meza-Figueroa et al., 2007). Furthermore, it is essential to note that the HMs present in dust have the potential to enter the human body via diverse routes, such as ingestion, inhalation, and dermal contact (Cao et al., 2015; Kang et al., 2011; Rout et al., 2013). Children are particularly susceptible to harmful substances in indoor dust due to their behavior, such as crawling, participating in activities that include hand-mouth contact, and having a rapid pace of growth (Beamer et al., 2008; Moya et al., 2004). Furthermore, Olujimi et al. (2015) discovered that swallowing dust is the most common exposure pathway to HMs in children. Since they play on the floor, children are exposed to dust through the digestion exposure pathway. According to the International Agency for Research on Cancer (IARC), Aluminium (Al), copper (Cu), iron (Fe), and zinc (Zn) are classified as non-carcinogenic. The carcinogenic and non-carcinogenic effects of arsenic (As), cadmium (Cd), chromium (Cr), nickel (Ni), cobalt (Co), and lead (Pb) are known (Sanborn et al., 2002; Tchounwou et al., 2003). Arsenic is capable of causing harm to the human body through its detrimental effects on the cardiovascular, digestive, and DNA systems (Sijko and Kozłowska, 2021). Lead negatively affects kidneys and causes neurological disorders (Ferreira-Baptista and De Miguel, 2005). High levels of Ni can lead to higher risks of cancer and heart attacks, as well as build up in various parts of the body like bones, liver, kidneys, and aorta (Silvera and Rohan, 2007).

The Human Health Risk Assessment (HHRA) approach, created by the United States Environmental Protection Agency (USEPA), calculates the human health risk associated with exposure to contaminants. According to Luo et al. (2012), HHRA is composed of four essential components: hazard identification, exposure evaluation, dose-response evaluation, and risk characterization. Determining the risk for HM exposure through indoor dust can be achieved by collecting and evaluating data from previous investigations. This allows us to determine whether a specific exposure to HMs may increase the likelihood of harming human health. Exposure evaluation involves linking HM transmission destiny with source, exposure location, and receptor (Meza-Figueroa et al., 2007; Tong and Lam, 2000). Furthermore, the dose-response assessment addresses the extent of exposure to HMs and related negative health impacts. Finally, risk characterization involves compiling all the data collected from the earlier stages and then quantifying the potential human health risks. Similarly, the hazard quotient (HQ) values show non-carcinogenic risks, whereas the carcinogenic risks (CR) estimate the likelihood of additional cancer occurrence during a lifetime. If HQ values exceed 1 it signifies that human health risk related with the presence of HMs in dust exceeds the acceptable level of non-carcinogenic risk, which may lead to chronic diseases unrelated to cancer. However, CR levels that are higher than the permissible range of carcinogenic risk of 1×10^{-6}–1×10^{-4} indicate an increased probability of additional cancer occurrence among exposed individuals throughout their lifetime. Research on environmental exposure to HMs carried out in the past (Latif et al., 2014; Tahir et al., 2007; Yap et al., 2012), primarily focused on identifying sources of environmental HMs and analyzing their composition. These investigations provided little information on the human health risks of exposure to environmental HMs.

As there are various sources of outdoor air pollution that expose humans to heavy metals (HMs) the objective of the present study was to fill some gaps in the reporting of the influence that HMs in indoor and outdoor environments can have on human health. The detailed aims of this study were to (1) review the literature on HM concentrations in different types of exposure environments and (2) determine carcinogenic and non-carcinogenic risks related to exposure to HMs for human health in those environments. In addition, this assessment can be used as a resource when formulating a policy that emphasizes protecting human health and the environment.

2. Materials and methods

2.1. Protocol development

A systematic review was carried out following the Cochrane protocol (Higgins et al., 2019), which was already published and entered into the International Prospective Register of Systematic Reviews (PROSPERO) (registration number CRD42023472427). The PRISMA procedure was employed to analyze the collected data (Liberati et al., 2009). A conceptual diagram of the assessment of environmental exposure to HMs investigated in this systematic review was presented in Fig. 2.

The PECO (Population, Exposure, Comparator, and Outcome) statement was utilized to address the particular goals of the systematic review before the search strategy was created. The particular stages of conducting the presented systematic review are shown in Fig. 3. This study consisted of eight steps: selecting the topic, developing the protocol and conceptualizing it, drafting the PECO statement, conducting a systematic literature search, selecting the study (screening and data extraction), evaluating the possibility of bias, clearing up any contradictions or ambiguities, synthesizing the findings, and interpreting the findings. The reviewed articles were obtained by searching the Web of Science, Scopus, PubMed, Embase, and Medline databases without a time limitation until December 30, 2023.

2.2. PECO statement

Developing a protocol constitutes a crucial element in the preliminary phase of the systematic review procedure. A PECO (Population, Exposure, Comparator, and Outcome) statement guides the development of the protocol. The structure of the inclusion and/or exclusion criteria and the literature search was laid out using the following identifiers as the basis (Morgan et al., 2018).

2.2.1. Population types

In our study, all age groups in both the formal and informal sectors were eligible for inclusion in the study. Also, studies did not include occupational exposure in any industrial setting.

2.2.2. Exposure types

We comprised studies that defined values of carcinogenic and non-carcinogenic risk due to environmental exposure to HMs in indoor and outdoor spaces.

2.2.3. Comparator types

The present investigation involved a comparison between the HM concentrations derived from the literature review and the acceptable limits established by the World Health Organization (WHO).

2.2.4. Outcome(s) types

Various indoor and outdoor environments under investigation were prioritized according to the degree of HM exposure determined by the HHRA analysis.

2.3. Literature sources and search strategy

The systematic review was conducted by searching for articles in online electronic databases, including the Scopus, Web of Science, PubMed, Embase, and Medline databases based on Medical Subject Headings (MeSH) until December 30, 2023. The search terms used were 1) risk OR hazard OR "risk assessment" OR "carcinogenic risk" OR "non-carcinogenic risk" OR "cancer risk" OR "non-cancer risk" OR "lifetime cancer risk" OR "hazard quotient" OR "HQ" OR "hazard index" OR "HI" OR "hazard assessment" OR "human health risk" OR "HRA"; 2) "outdoor environment" OR "outdoor air" OR "indoor air" OR "indoor environment"; 3) exposure OR "inhalation exposure" OR "respiratory exposure"; and 4) "heavy metal*" OR "trace element*" OR "trace metal*".

Fig. 4 presents a revised PRISMA flow chart summarizing the study selection process and its results at each stage. The selection of articles was performed in two screenings. In the initial screening, 407 articles were included after the analysis of the titles and abstracts. The full texts of these articles were then extensively studied in the second screening to ensure that they met the requirements, such as research focusing on children and study locations. The 407 articles were subjected to a second screening procedure, resulting in the selection of 43 studies that were deemed relevant. Of these 43 articles, the contents of HMs in environments of indoor and outdoor were obtained and their concentrations were presented in this systematic review in $\mu g\ m^{-3}$.

2.4. Inclusion and exclusion criteria

The present systematic evaluation exclusively considered articles that satisfied the following: the full text was accessible, the investigation focused on the global impact of heavy metals (HMs) on environmental contamination (analyzed global environmental pollution caused by HMs), and original articles published in peer-reviewed journals in English. Furthermore, we did not include the following in our study: books, review- papers, presentations, republished data, conference papers, letters to the editor, articles without an available full text, articles on the development of methods to HM analysis, articles on the HM concentrations in occupational exposure, articles that did not report non-carcinogenic (HQ) and carcinogenic (CR) risk values.

2.5. Data collection

The articles were evaluated according to predetermined inclusion and exclusion criteria, focusing on the relevance of their titles and abstracts. The full texts of the eligible articles were then independently reviewed by several reviewers, and the data was extracted and recorded in the data collection form as follows: first author, year of publication, country, point of sampling, research duration, sample size, detection device, HM type, concentration (mean), and values of CR and HQ.

2.6. Quality control

The quality assessment of the chosen articles was carried out by two researchers (A.H.Kh and S.Gh) using the Joanna Briggs Institute (JBI) checklist (Moola et al., 2017). This checklist includes 22 parts that assess various aspects of methodology, such as sampling techniques, variable measurement, appropriate statistical analysis, exposure assessment, confounding variables, results, and study objectives. The authors adopted a simple scoring method. Each part of the checklist questions provided possible answers as follows: "yes", "no", "uncertain", or "inapplicable". Two researchers compared the answers given to the articles.

The quality of the articles was classified into three categories using this Q index for the different perceptions of quality: high-quality level (Q_1), medium quality level (Q_2), and low quality level (Q_3). If the answers to >50 % of the questions were "yes," the article was of high quality and had a low risk of bias. In contrast, if <50 % of the queries

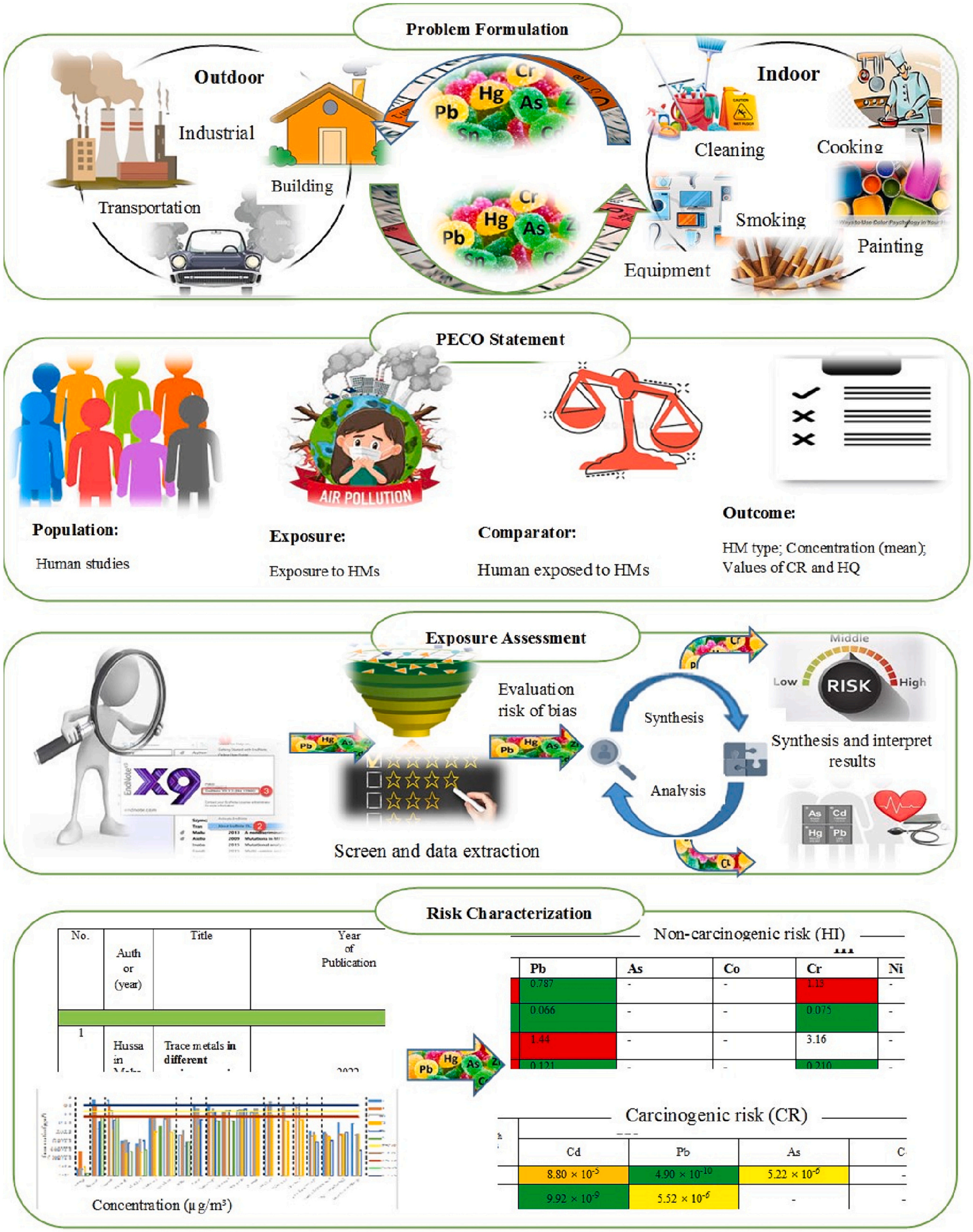

Fig. 2. Conceptual diagram of the steps of the systematic review on HHRA of exposure to HMs in the indoor environments.

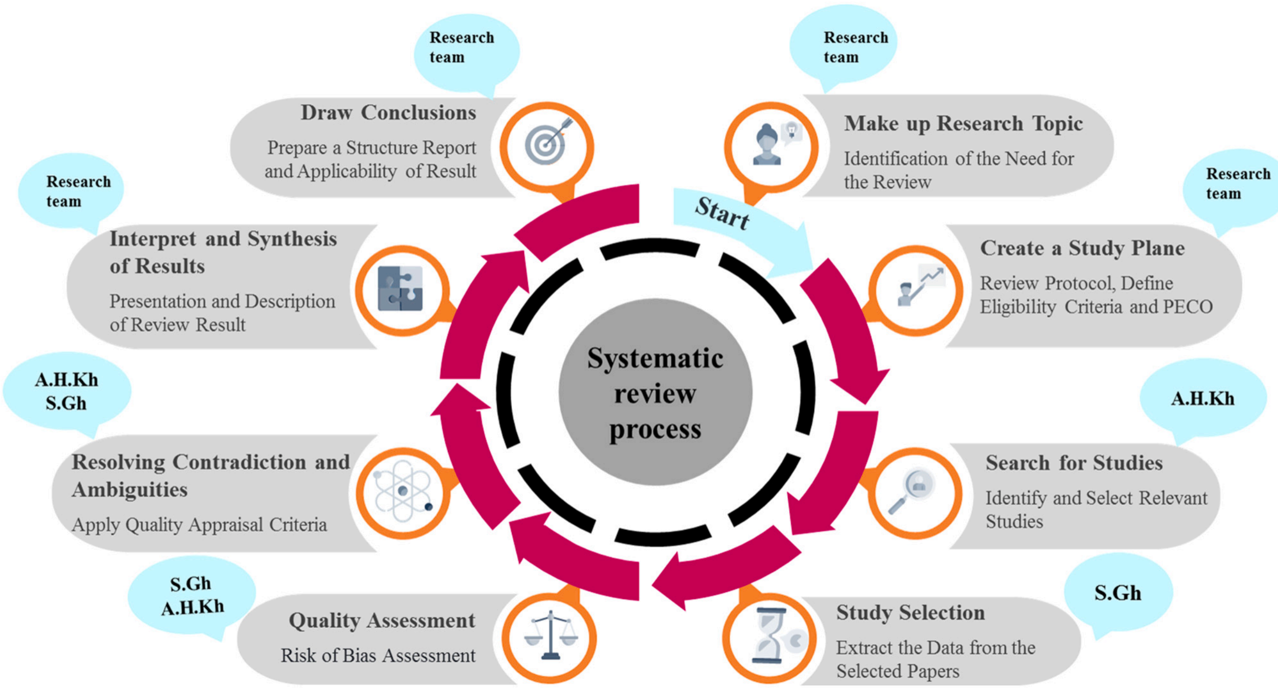

Fig. 3. Visual representation of the eight stages of the systematic review in this study.

were answered with "no," the article was of low quality and carried a high risk of bias. Furthermore, if 50 % of the questions had "unclear" responses, there was an unclear risk of bias, and the quality of the article was also rated as medium (Yu et al., 2020). All selected articles were accepted for further research due to their classification as high quality or low risk of bias after quality review.

2.7. Human health risk assessment

The articles included in this study used the USEPA health risk assessment to calculate carcinogenic (CR) and non-carcinogenic (HQ) risk values. A brief description of this method is given below.

The inhalation average daily dose (ADD_{inh}) values were calculated using Eq. (1) (USEPA, 2004; USEPA, 2011):

$$ADD_{inh} = \frac{C \times EF \times ED \times IR}{PEF \times AT \times BW} \quad (1)$$

Where ADD_{inh} is the average daily dose (mg kg^{-1} day^{-1}), C is the HM concentration in the ambient air (mg kg^{-1}), EF is the exposure frequency (days $year^{-1}$), ED is the exposure duration (years), IR is the intake rate of HMs (mg day^{-1}), PEF is the particle emission factor (m^3 kg^{-1}), BW is the body weight (kg), and AT is the averaging time (ED × 365 days). The parameter values needed for health risk calculations are provided in the supplementary material (Tables S1 and S2).

The non-carcinogenic (HQ) risk values were calculated using Eq. (2) (USEPA, 2011):

$$HQ = \frac{ADD_{inh}}{RfD_{inh}} \quad (2)$$

Where HQ is the hazard quotient (unitless), and RfD_{inh} is the reference dose specified for each HM (mg kg^{-1} day^{-1}). The RfD values were obtained from the USEPA Integrated Risk Information System (USEPA, 2011).

If the HQ values were ≥ 1, the non-carcinogenic risk was defined as unacceptable. If the HQ values were < 1, the risk was defined to be at the acceptable level (USEPA, 1989) (Table 1).

The Hazard Index (HI) describes the total health risk arising from exposure to several investigated compounds. In our case, the HI values determined the total risk of the inhalation exposure pathway to the investigated heavy metals and were calculated according to Eq. (3). If the HI values are <1 it is unlikely that harmful health impacts will manifest. When the HI values are ≥ 1 a considerable non-carcinogenic risk is stated that can lead to adverse effects on human health:

$$HI = \sum^{n} HQ_i \quad (3)$$

Where HI is the Hazard Index determined by summing the HQ values of all the examined HMs in the study.

The International Agency for Research on Cancer (IARC) distinguished five groups of carcinogens based on the certainty level of evidence in causing additional cases of cancer: Groups 1 (human carcinogenicity), Group 2A (probably human carcinogenicity), Group 2B (possibly human carcinogenicity), Group 3 (not classifiable as human carcinogenicity), and Group 4 (probably not human carcinogenicity). According to IARC, Cd, Cr, and As from Group 1 (proven human carcinogens) and Ni, Co, and Pb from Group 2 (possible human carcinogens) are considered carcinogenic elements for humans. Other investigated HMs, such as Al, Co, Cu, Fe, Ni, and Zn, are not considered carcinogenic. In this review, the carcinogenic risk due to inhalational exposure to HMs in air dust was investigated and the CR values were calculated according to Eq. (4):

$$CR = ADD_{inh} \times SF_{inh} \quad (4)$$

Where CR is carcinogenic risk (unitless), and SF is the cancer slope factor (mg kg^{-1} day^{-1}) for particular HM. The SF values were obtained from the USEPA Integrated Risk Information System (IRIS) database (USEPA, 2011).

The acceptable level of carcinogenic risk can be stated as a value in the range of 1×10^{-6} to 1×10^{-4}. If the CR values are below the acceptable level, the carcinogenic risk is negligible. When the calculated CR values exceed the acceptable level, additional cases of cancer are likely to occur in the exposed population (Table 2).

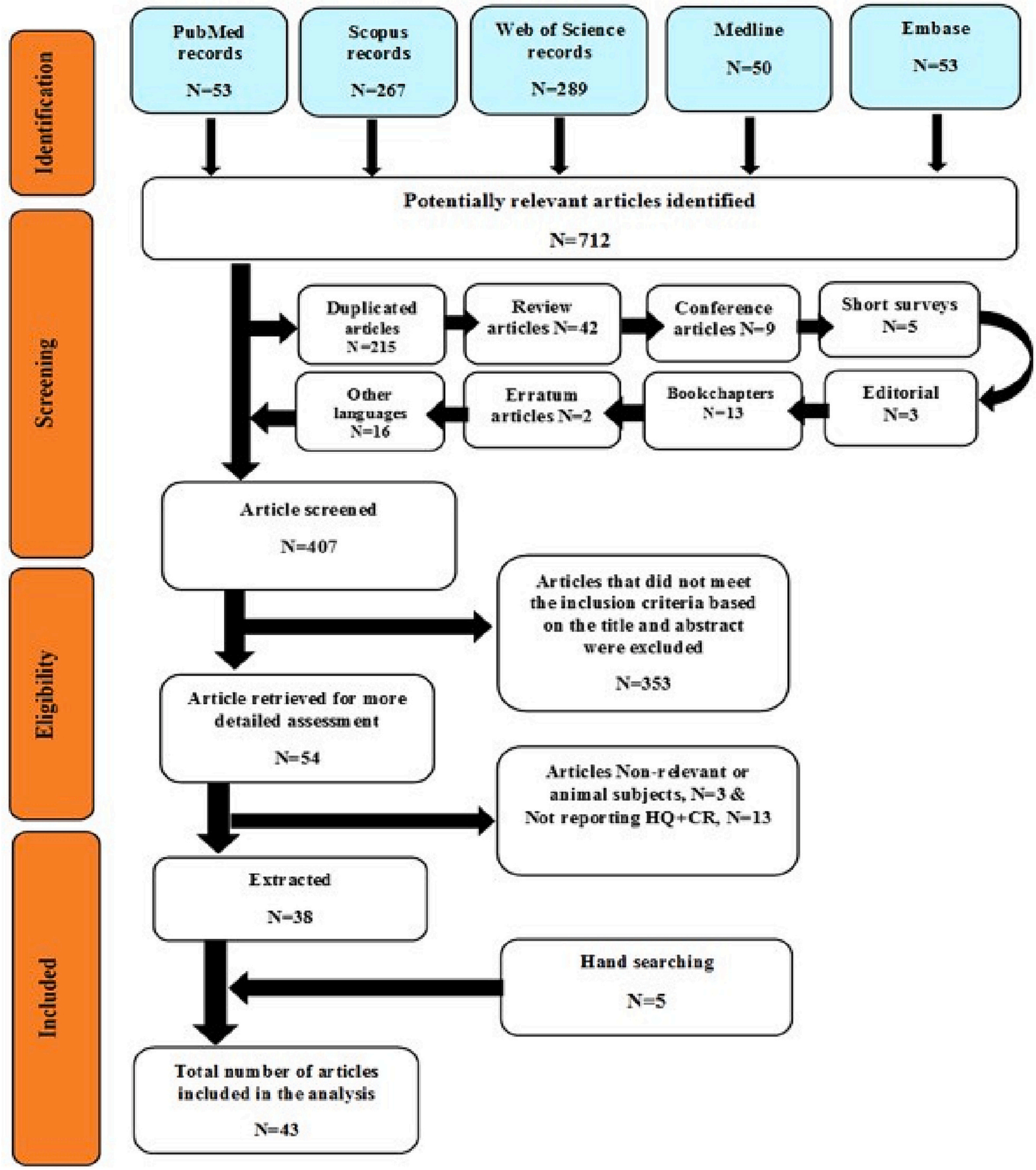

Fig. 4. PRISMA flow chart of article selection performed in the systematic search.

Table 1
Non-carcinogenic risk (HQ) levels based on values provided by the WHO.

Non-carcinogenic risk level	HQ
Acceptable risk	HQ <1
Potential to result in adverse effects risk	HQ ≥1

Table 2
Carcinogenic risk (CR) levels based on values provided by the WHO.

Carcinogenic risk level	CR
Definite risk	$10^{-4} < CR$
Probable risk	$10^{-5} < CR < 10^{-4}$
Possible risk	$10^{-6} < CR < 10^{-5}$
Negligible risk	$CR < 10^{-6}$

2.8. Synthesis and analysis of results

Meta-analysis as a quantitative synthesis in the study was not feasible due to the methodological and contextual heterogeneity. As a result, the average HM concentrations and risk estimates for the carcinogenic and non-carcinogenic effects of HM exposure were correctly integrated from the chosen studies (Tables 4–9). In this study, narrative synthesis was carried out in two stages: (1) the first stage was based on the general classification of articles according to the type of environment, and (2) a comparison of the findings within and between the studies of each group of environments with respect to the HM concentrations and the evaluated health risk values.

3. Result and discussion

3.1. Systematic search results

As a result of the review of scientific databases, 712 articles were initially obtained and subsequently 305 books, presentations, republished data, conference papers, and letters to the editor were eliminated. The authors performed a full-text eligibility assessment on the remaining 407 records, and 353 articles were found eligible. After reviewing the full texts, 54 articles matched the inclusion and exclusion criteria. At this stage, 38 studies were selected, while the remaining 16 articles were eliminated for the following reasons: 1) 3 articles were unrelated or applied to animals, and 2) 13 articles did not report the HQ and/or CR values. Furthermore, a hand searching was conducted to identify any further potential studies that met the inclusion criteria for the systematic survey. Ultimately, five additional studies were acquired by manual search, increasing the total number of articles included in the present investigation to 43. Fig. 4 presents a revised PRISMA flow chart summarizing the study selection process and its results at each stage.

According to the origin of the investigations included in the systematic review, they were conducted in several countries: sixteen in China, eight in Malaysia, four in Iran, four in India, three in Saudi Arabia, three in Turkey, two in Nigeria, and after one study in each Portugal, South Africa, Poland, and Kuwait. Regarding the environment of the studies performed, the 43 studies mentioned above were grouped regarding the environment as follows: household environments, educational environments, office and commercial environments, and other types of environments. The standard limits imposed by the USEPA National Ambient Air Quality Standards (NAAQS), the World Health Organization (WHO), and the European Union (EU) are presented in Table 3.

3.2. HM concentrations in household environments

Our research investigated 16 articles on HM concentrations in household environments (Table 4). China was revealed as the country with the highest number of studies on HM exposure in this environment, with approximately 53.3 % of the articles included in this systematic review coming from this country. Other countries with research activity in this field were Saudi Arabia, Kuwait, South Africa, India, and Iran.

According to the investigations, the most common analyzer used to measure the concentration of HM in air samples was Inductively Coupled Plasma-Mass Spectrometry (ICP-MS). Other methods used in these studies to determine HM concentrations were Inductively Coupled Plasma-Optical Emission Spectrometry (ICP-OES), Atomic Absorption Spectrometry (AAS), and Anodic Stripping Voltammetry (ASV) (Table 4). As presented in Fig. 5, the analyzed studies focused on the concentrations of As, Cd, Pb, Ni, Co, Cr, Zn, Cu, and Mn, which are considered the most hazardous among the elemental pollutants (Mishra et al., 2019).

In most of the studies conducted in the household sector, the highest concentrations were related to zinc, copper and manganese elements. Only three of the 16 studies (Albar et al., 2020; Lion and Olowoyo, 2021; Yang et al., 2015) examined in the household sector found concentrations of As, Cd, Ni, Pb, Cr, and Co lower than the limit values set by the WHO (Organization, 2000), the EU (Europe, 2005), and the NAAQS (NAAQS, 2020) (Fig. 5). The concentrations of Zn, Cu, and Mn in the studies of Albar et al. (2020), AlMulla et al. (2022), Li et al. (2022a), Rabha et al. (2018), Li et al. (2016), Hashemi et al. (2020), and Lin et al. (2015), were higher than the limit values set by the World Health Organization (WHO) (Europe, 2000), the European Union (EU) (Europe, 2005), and the National Ambient Air Quality Standards (NAAQS) (NAAQS, 2020) (Fig. 5).

If detected concentrations of HMs exceed the specified threshold levels, it is assumed that indoor environment pollution is caused by anthropogenic activities (Yongming et al., 2006). Pb concentrations were likely to exceed limit values due to activities in the industrial sector and traffic emissions, as observed in European urban sites (Grivas et al., 2018; Titos et al., 2014). Considering the impact of outdoor air conditions on indoor air quality, aerosols and pollutants can infiltrate indoor spaces, exacerbated by indoor human activities. In contrast, the infiltration rates of fine particles, including carbonaceous aerosols and HMs originating from industrial and urban emissions, are greater than those of coarse dust particles from the desert, owing to the latter's diminutive dimensions (Liu et al., 2003). In general, differentiation in element concentrations can lead to the definition of emission sources of dust particles and HMs observed in household environments (Al Hejami et al., 2020). In addition, Co, Cr, and Ni concentrations may serve as indicators of a geogenic emission source, including meteorological processes and the weathering of materials within the Earth's crust (Aghadadashi et al., 2019). The non-normal distribution of elements indicates the influence of multiple factors on the composition of indoor dust particles. The concentrations of Ni, As, Cr, and Fe in the larger dust fractions indicate local soil sources, while the content of Pb indicates its anthropogenic emissions. However, fine dust particles transport HMs better due to their increased specific surface area and clay minerals and organic materials (Yao et al., 2015).

Proximity to main roads can influence variations in HM concentrations between houses (Baccarelli et al., 2009; Wang et al., 2017). However, differences in indoor human activities are the most influential factor in changes in HM concentrations in household environments. Varying periods of natural ventilation through the opening of windows in dwellings can significantly vary the concentrations of HMs. In addition to open windows and doors, inadvertent building penetration and mechanical ventilation systems can allow outdoor air contaminants to enter the house. Thus, the materials chosen, building air permeability, and design regulations for naturally ventilated structures are crucial

Table 3
Limits values of HMs in ambient air (μg m^{-3}) according to WHO, NAAQS, and EU air quality guidelines.

Source	Pb	Cd	As	Ni	Cr	Co	Zn	Mn	Cu
μg m^{-3}									
WHO[a] (2000) (Organization, 2000)	0.5	0.005	0.0066	0.025	0.00025	0.002	–	0.150	–
NAAQS[b] (2020) (NAAQS, 2020)	1.5	0.003	0.01	0.020	0.012	–	1	0.2	100
EU[c] (2005) (Europe, 2005)	0.5	0.005	0.006	0.020	–	–	–		–

- Not defined.
[a] World Health Organization.
[b] National Ambient Air Quality Standards.
[c] European Union.

Table 4
The characteristics of studies on HM exposure in household environments.

Country	Sampling environment	Analysis method	Duration of sampling	Point of sampling	Sample size	References
Saudi Arabia	Households	ICP-MS	–	Jeddah (urban area)	10	Albar et al., 2020
Kuwait	Kitchen (indoor houses)	ICP-MS	12 months	Kuwait	50	Al-Harbi et al., 2021
Saudi Arabia	Indoor dust sampling	ICP-OES	3 months	Al-Qatif,	9	AlMulla et al., 2022
South Africa	Household (1 outdoor &1 indoor)	ICP – MS	–	Brits	40	Lion and Olowoyo, 2021
China	House dust	Atomic absorption spectrometry	3 months	Jinzhong	–	Yang et al., 2015
China	indoor- outdoor personal samples	Atomic fluorescence spectrome- try	9 months	Heating period	60	Li et al., 2022b
China	Residential building	ICP-OES	6 months	Changchun	–	Li et al., 2022a
China	Indoor environments including home, office, cyber-classroom, and chemical analysis room	ICP – MS	_	Guangzhou	11	Lu et al., 2019
China	Homes (outdoor and indoor)	ICP-MS	3 months	Pearl River Delta (Guangzhou, Shenzhen, Dongguan, Foshan, Zhongshan, Zhuhai, Huizhou, Zhaoqing, Jiangmen)	22	Wu et al., 2019
China	Apartment	ICP-MS	3 months	Kitchen	16	Li et al., 2021
India	Houses in the urban microenvironment	ICP-AES	3 months	Agra	54	Rohra et al., 2018
China	Bedrooms and living rooms of homes	ICP-MS	6 months	Chengdu	63	Li et al., 2016
Iran	Private dwelling units	ICP-MS	3 months	Asaluyeh	40	Tashakor et al., 2022
China	Residential floor	ICP-OES	_	Anhui	25	Lin et al., 2015
India	Households with separate kitchens	Anodic Stripping Voltmeter	3 months	Assam	200	Rabha et al., 2018
Iran	Home	Atomic Absorption Spectrometry	6 months	Bushehr	19	Hashemi et al., 2020

–Not defined.

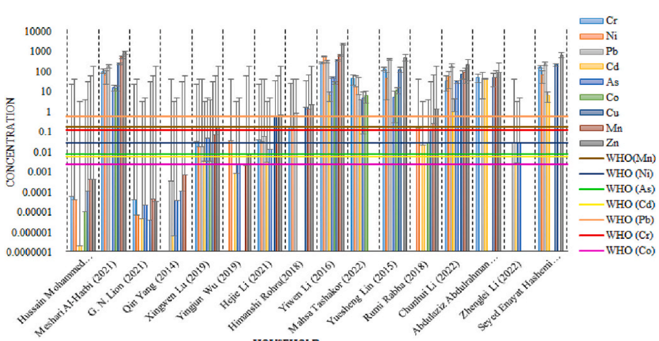

Fig. 5. Comparisons of HM concentrations in household environments (μg m^{-3}).

variables in controlling indoor air pollution levels (Riley et al., 2002; Tian et al., 2009). Moreover, the quantity of inhabitants in each residence can play a crucial role in assessing indoor HM levels (Kurt-Karakus, 2012). Some studies reported higher concentrations of Fe and Zn in houses with only 1 to 2 residents (Chattopadhyay et al., 2003). The differences in heavy metal concentrations across households may also be influenced by the functioning of the air conditioning system, which was linked to higher levels of Cu, Cr, Cd, and Mn in Istanbul, Turkey (Kurt-Karakus, 2012). Studies have shown that wall paint can contain high levels of HMs. Green paint is linked to elevated levels of Cu, purple paint to Zn and Pb, and yellow paint to Cd, Cu, Pb, and Zn (Kurt-Karakus, 2012; Tong and Lam, 2000). Consequently, with the time quality of the wall paints used due to regulatory factors in terms of the HM content, it could improve and affect the variability in concentrations between houses depending on the last time of wall painting.

The influence of the family has also been confirmed to have an impact on the air concentrations of HMs (Zhou et al., 2020). To the most important internal factors that affect indoor HM concentrations belong the type of fossil fuel combustion and heating at the house, the type of floor covering, the type and color of wall paint, the age of the building, and the smoking habits of the residents (Gul et al., 2023). Cao et al. (2024) reported that the health risk of HM concentrations in household dust increased with decoration within 5 years, purchase of new furniture within 1 year, increase of the residence duration, and with increasing housing age. Among external factors affecting outdoor air quality, the following sources are mentioned: mining, traffic emissions, agricultural and industrial activities, and natural sources (Somsunun et al., 2023). In relation to the above, the customs of residents of opening doors and windows in order to ventilate the rooms significantly affected the air exchange between indoor and outdoor environments increased due to high-frequency windowing (Cao et al., 2024).

Children were stated to be more susceptible to exposure to hazardous materials (HMs) because of their physiological and behavioral traits related to their physical activity and the time they spend playing on playgrounds (de Burbure et al., 2006; Rangel-Mendez et al., 2016). Moreover, due to children's frequent hand-to-mouth activities and larger relative breathing volume of polluted air to body weight, they are also more susceptible than adults to indoor air pollution, particularly in the inhalation and ingestion pathways (Faustman et al., 2000). HMs' toxicity, bioaccumulation, and restricted degradation capacity in humans present significant health risks (Gao and Ji, 2018). To summarize, the risk value resulting from inhaling particulate matter was deemed unacceptable in over 70 % of the examined samples within the residential settings group since the HM contents were significantly higher than the WHO-recommended limit value.

3.3. HM concentrations in educational environments

In our research, 16 articles on HM concentrations in educational environments were considered (Table 5). Regarding the country, these

Table 5
The characteristics of studies on HM exposure in educational environments.

Country	Sampling environment	Analysis method	Duration of sampling	Point of sampling	Sample size	References
School						
Malaysia	Primary school	ICP-MS	9 months	City center	51	Mohamad et al., 2016
Malaysia	Indoor and outdoor classrooms	ICP-MS	6 months	Kuala Lumpur (Indoor)	44	Othman et al., 2019
Malaysia	Indoor and outdoor school areas	ICP-MS	9 months	Kuala Lumpur (Indoor)	228	Othman et al., 2022
Malaysia	Indoor classroom	(ICP-MS)	3 months	Kuala Lumpur (Indoor)	–	Alias et al., 2021
Saudi Arabia	primary schools	ICP- OES	3 months	Jeddah	40	Alghamdi et al., 2019
Nigeria	Schools	ICP-OES	3 months	Lagos	–	Famuyiwa and Entwistle, 2021
Nigeria	Nursery and primary school classrooms	Microwave Plasma Atomic Emission Spectrometry (MP-AES)	3 months	Abeokuta	20	Famuyiwa et al., 2023
Iran	Laboratory rooms; school classrooms	Atomic Absorption Spectrometry	3 months	Bushehr	27	Hashemi et al., 2020
Poland	Kindergartens located in and near city	Atomic absorption spectroscopy (AAS)	3 months	Gliwice	–	Mainka, 2021
University						
Malaysia	Lecture room, laboratory, and student hostel room (Hostel)	Atomic absorption spectrometry (AAS)	3 months	Lecture room	192	Sulaiman et al., 2017
China	Campus of University (Indoor and outdoor)	ICP- OES	3 months	Nanjing	52	Wang et al., 2019
Iran	School of University (indoor environment)	ICP- OES	3 months	Ardabil	10	Vosoughi et al., 2021
China	Indoor cyber-classroom, and chemical analysis room	ICP-MS	_	Guangzhou	88	Lu et al., 2019
Turkey	University (indoor- second floor)	(ICP- MS/MS)	3 months	Gölköy bolu	30	Arı, 2020
China	University dormitory	Atomic Ab- sorption Spectrophoto meter	6 months	Lanzhou	13	Bao et al., 2019

- Not defined.

studies were carried out in Malaysia, China, Saudi Arabia, Nigeria, Iran, Poland, and Turkey. Malaysia was revealed as the country of the highest number of HM exposure research in this environment.

3.3.1. School environments

In ten studies, HM concentrations were investigated in school environments (Alghamdi et al., 2019; Alias et al., 2021; Famuyiwa et al., 2023; Famuyiwa and Entwistle, 2021; Hashemi et al., 2020; Mainka, 2021; Mohamad et al., 2016; Othman et al., 2019; Othman et al., 2022; Tan et al., 2018), and in six studies in universities (Arı, 2020; Bao et al., 2019; Lu et al., 2019; Sulaiman et al., 2017; Vosoughi et al., 2021; Wang et al., 2019).

Airborne HMs concentrations in these environments were determined using Inductively Coupled Plasma-Optical Emission Spectrometry (ICP-OES), Inductively Coupled Plasma-Tandem Mass Spectrometry (ICP-MS/MS), Inductively Coupled Plasma-Mass Spectrometry (ICP-MS), Microwave Plasma-Atomic Emission Spectrometry (MP-AES), and Atomic Absorption Spectrometry (AAS). HM concentrations in schools

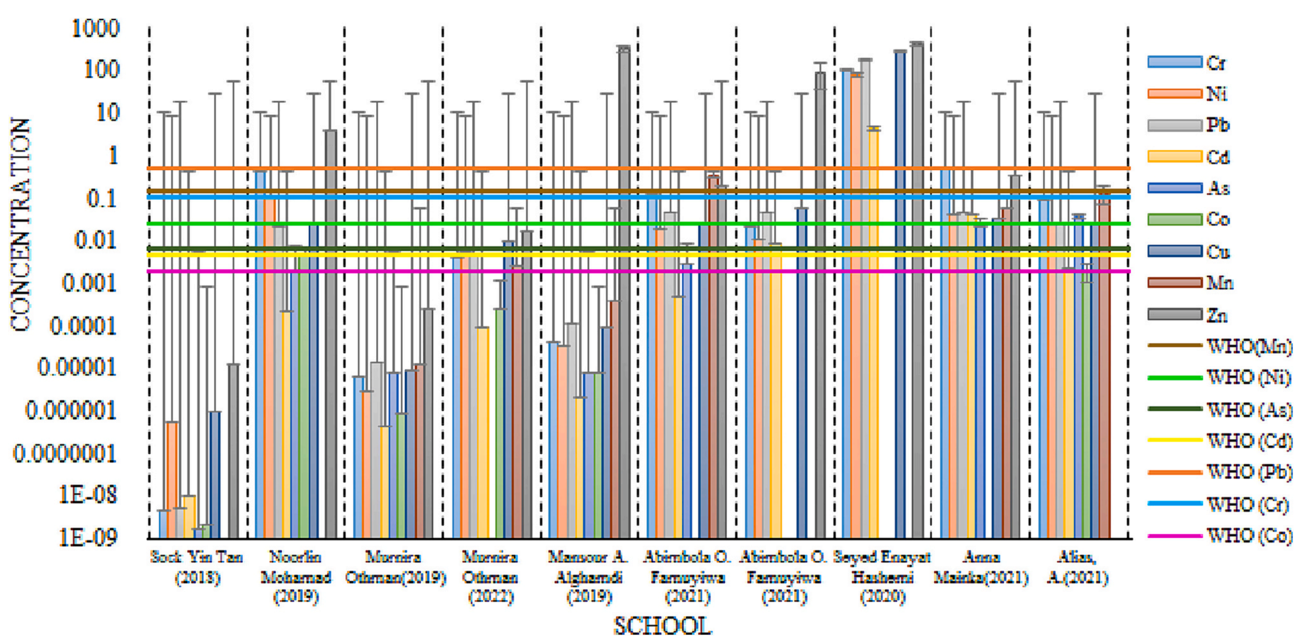

Fig. 6. Comparisons of HM concentrations in school environments (μg m^{-3}).

varied by location (Table 5).

As presented in Fig. 6, the concentrations of Cd, Cr, Ni, and Pb found in schools in Bushehr (Iran) (Hashemi et al., 2020), in kindergartens in Gliwice (Poland) (Mainka, 2021), Cr, As, and Ni in schools in Malaysia (Alias et al., 2021; Mohamad et al., 2016), and Cr and Mn in schools in Lagos (Nigeria) (Famuyiwa and Entwistle, 2021), were higher than the limit values established by the WHO, the EU, and the NAAQS. In classrooms located in Iran, the decreasing order of the mean HM values was following: Cu > Zn > Pb > Cr > Ni > Cd (Hashemi et al., 2020). The HM concentrations in classrooms in Gliwice, Poland, were the highest for Cr (0.495 μg m^{-3}) followed by Zn (0.3625 μg m^{-3}), Mn (0.0565 μg m^{-3}), Pb (0.04625 μg m^{-3}), Ni (0.042 μg m^{-3}), Cd (0.04125 μg m^{-3}), Cu (0.0325 μg m^{-3}), and As (0.0275 μg m^{-3}) (Mainka, 2021).

In classrooms, fans help circulate air, while opening windows allow dust from outside to be carried inside by the wind (Atkinson et al., 2009). Contaminated dust carried by the wind can enter classrooms through natural ventilation, such as open windows and doors. The dust is then deposited on desks, windows, floors, and chairs, exposing children to HMs from the dust through inhalation, ingestion, and dermal contact (Latif et al., 2014). Turner (2011) and Yuswir et al. (2013) stated in their studies that breathed dust by children can enter their digestive systems, including the stomach and intestines, and cause HM to be absorbed and circulated throughout the body via the bloodstream. Road dust was found to be transferred to indoor classrooms by schoolchildren because of the high factor load of Al, Fe, Co, Zn, and As in the study by Othman et al. (2019).

The elevated levels of pollution were stated in the vicinity of the school resulting from several industrial operations. Cd, which is frequently employed as a cigarette marker in indoor contexts, was utilized to identify the effect of cigarette smoke emissions (Na et al., 2004; Slezakova et al., 2009). Although smoking is prohibited in schools, the transfer of cigarette smoke from smokers may affect the quality of the indoor air. According to the studies conducted by Slezakova et al. (2009) and Hunt et al. (2011), the fine fraction of cigarette smoke is a major cause of indoor pollution. According to the results of Famuyiwa and Entwistle (2021), HM emissions from indoor schools in Lagos were caused by construction materials (Al, As, Fe, Mn, Ni, and U), human activities (Cd, Cu, and Pb), and the combination of these two activities (Cr, V, and Zn).

The issue of traffic pollution surrounding elementary and secondary schools has garnered considerable attention, with numerous experts and scholars focusing on reducing the risk that traffic pollutants pose to students' health. Previous research has demonstrated that pollution in the vicinity of schools impacts the neurologic, pulmonary, and cardiovascular systems of the students and their cognition. The infiltration of emissions from outdoor pollution sources, such as traffic, vehicle repair, waste incineration, playground dust, and other sources, from outside pollution sources, significantly contributes to dust within educational institutions. However, urban road dust is a mixture of non-vehicle exhaust pollutants (such as tire and brake wear) and road pavement, which is linked to road traffic. While Pb, Zn, and Cu are prevalent in vehicle emissions on the road, their toxicity depends on the route and exposure dose. As a result, high-traffic regions frequently have higher levels of heavy metals in road dust. Inadequate vehicle maintenance, frequent pauses, and traveling at speeds below 30 to 40 km h^{-1} can contribute to increased fuel consumption, which is harmful to human health and the environment. According to Madureira et al. (2012) road transport emissions could be a notable source of metal exposure, such as Cr, Cu, Pb, and Zn, in school environments.

3.3.2. University environments

The levels of investigated HMs in indoor dust samples taken from universities are presented in Fig. 7.

Concentrations of As, Cd, Cr, Co, Ni, and Pb in indoor dust from studies of Wang et al. (2019), Vosoughi et al. (2021), and Bao et al. (2019) were much higher than in indoor dust reported by Sulaiman et al.

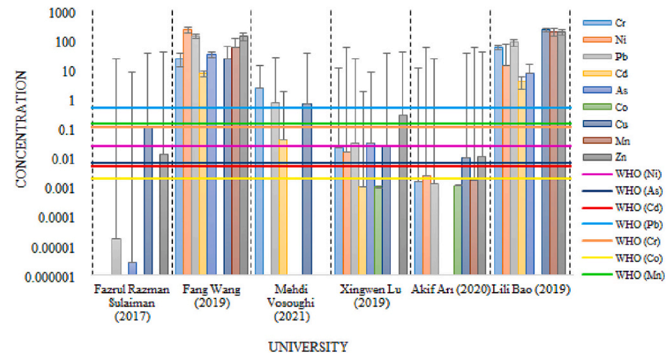

Fig. 7. Comparisons of HM concentrations in university environments (μg m^{-3}).

(2017), Lu et al. (2019), and Arı (2020). These concentrations were higher than the limit values set by the WHO, the EU, and the NAAQS. Being exposed to indoor pollutants can affect student performance (Mohai et al., 2011). Zhong et al. (2014) in their study suggested that university students' productivity and health may be affected by indoor air quality.

The origins of indoor air pollutants can vary depending on the activities of the occupants, the location of a building, indoor sources, such as emissions from building materials, and outdoor sources, such as fuel combustion (Mohamad et al., 2016). Pb in classroom dust can come from interior sources such as construction materials, cleaning and hygiene items, computers, printers, or external sources such as outdoor particles. Many distinct and well-known industrial and consumer goods contain Ni; however, in recent years, mobile phones have emerged as a possible source of Ni in educational settings (Klein and Costa, 2022). Zn and Pb from human activities may also come from the utilization of Zn—Pb based paint (McAlister et al., 2008; Sulaiman et al., 2016). Corrosion of Zn-coated items within the sample facility could cause Zn to be present in the indoor dust composition (Zhong et al., 2014). The elevated Zn concentration in indoor dust is likely attributed to the presence of suspended dust particles originating from both outdoor and indoor activities, especially in the lab building.

The earth's crust and the movement of people into and out of buildings have the most effects on the composition of indoor dust when it comes to metals. Compared to working adults, students at schools and universities could spend more time indoors (Ashmore and Dimitroulopoulou, 2009; Raysoni et al., 2013). Vehicle emissions contribute primarily to Cd, Cr, Cu, Ni, Pb, and Zn (Al-Rajhi et al., 1996; Hassan, 2012). Indoor dust has more diversified sources of origin and a higher particle concentration than outdoor dust. Therefore, the toxicity and composition of indoor dust are more complex, endangering human health to a greater extent (Martins et al., 2020). Outdoor environments were recognized as a primary source of indoor pollutants, since they collect contaminants from their migration in air, water, and soil (Srivastava and Jain, 2007). In addition to infiltration from outdoor spaces, indoor dust sources include cooking, cleaning, people's movements, painting, and furnishing (Abt et al., 2000). Particles in indoor air absorb air contaminants, which eventually settle as dust (Latif et al., 2014). Braniš et al. (2005) in their studies found that ventilation and human mobility affected indoor air particles in buildings. Small particles with low deposition velocities may linger in the air for extended periods, possibly affecting human respiration (Matson, 2005). Zhou et al. (2019) reported that levels of HM in indoor PM were higher than in outdoor pollution, implying poor indoor air quality and related carcinogenic risk.

3.4. HM concentrations in office and commercial environments

In our research 4 articles investigated HM concentrations in office environments (Lu et al., 2019; Massey et al., 2016; Othman et al., 2016, 2018) and 4 articles in commercial centers, such as restaurants (Dai et al., 2018; Taner et al., 2013), tobacco cafés (Masjedi et al., 2019), and shopping centers (Massey et al., 2009) (Table 6).

Airborne HM concentrations in these working environments were quantified using Inductively Coupled Plasma-Mass Spectrometry (ICP-MS), Atomic Absorption Spectrometry (AAS), and Energy Dispersive X-ray Fluorescence (ED-XRF) Spectrometry.

In office-related studies, the concentrations of Pb, Ni, Cr, and Cd in various indoor microenvironments in India were higher than the limit values set by the WHO, the EU, and the NAAQS (Fig. 8). However, in the study carried out in Malaysia and China, Cr and As concentrations were higher than the limit values set by the WHO.

Vehicle emissions were identified as a contributing factor to the presence of HMs in indoor dust. Similarly, studies of Darus et al. (2012) and Latif et al. (2014) found that the predominant origin of HM in indoor dust in Malaysia was vehicle emissions. Population growth has led to an increase in traffic volume in Rawang, Malaysia. The Highway Planning Division of the Malaysian Ministry of Works gathered information in 2015 that showed that the yearly growth rate of Rawang road traffic has increased by 5.48 % (Malaysian Ministry of Works, 2014). The volume of road traffic increases with increasing number of vehicles. Vehicle emissions, such as engine oil usage, collisions with car components, tires, brake pads, and engine wear, were the sources of Cu, Cr, Ni, Pb, and Zn (Latif et al., 2014; Sulaiman et al., 2017). Additionally, the concentration of HMs in the indoor air of offices can be caused by smoking, traffic emissions, aged construction materials, construction colors, and various mechanical and electrical equipment in workplaces, such as computers, printers, and photocopiers (Arı, 2020; Hashemi et al., 2020).

The findings derived from the commercial environment evaluations (Dai et al., 2018; Masjedi et al., 2020; Massey et al., 2009; Taner et al., 2013) (Fig. 9) indicated that the levels of HMs in all the analyzed studies were above the WHO limit values.

In the Chinese restaurants investigated, Pb was the element with the highest concentrations (>60 % of the investigated restaurants), followed by Hg (20 %), Cr (7 %), Co (5 %), and Ni (4 %). HMs such as Pb could be attributed to the use of metallurgical cooking utensils and dining ware subjected to high temperatures (Dai et al., 2012). Anthropogenic activities, such as the cooking fuel combustion, may contribute to the release of metallic elements into the environment (Xu et al., 2012). Tobacco smoke was a significant contributor to the release of toxic substances into the interior air of cafes, and $PM_{2.5}$ was a substantial contributor to the accumulation of these substances (Gorka-Kostrubiec, 2015). The decreasing order of HM contents in the cafés studied in Tehran, Iran was Pb > Ni > Cr > Cd. Total concentrations (mean ± SD) of investigated HMs were: 1118.5 ± 50.42 ng m^{-3} for WCC (cigarette and water pipe cafés), 663.64 ± 40.79 ng m^{-3} for WC (water pipe cafés), 425.57 ± 17.55 ng m^{-3} for CC (cigarette cafés), and 79.02 ± 5.13 ng m^{-3} for SFC (smoking free cafés) (Masjedi et al., 2020). Tobacco plants accumulate nickel in their leaves after absorbing it from the soil, and as they are burned, Ni bound to PM is released into the interior air of cafés (Gorka-Kostrubiec, 2015). Investigations showed that the percentage of Cr in the five lobes of the lungs of smokers was much higher than that of non-smokers (Pääkkö et al., 1989). Therefore, the presence of Cr in the indoor air of the cafés examined poses a significant public health issue that requires attention. Some of the factors that affected the concentration of HMs included the type of tobacco used, the status of the heating systems (on/off), the number of consumers, the type of circulation system used, the airflow speed, the ambient temperature, the moisture content of the air, and the cafe's surface area (Masjedi et al., 2020).

3.5. HM concentrations in other types of environments

From the studies included in our research, 5 articles were assigned to other types of environments, related to service provider centers and urban outdoor environments. This group includes subway stations (Ji et al., 2022), hospitals (Slezakova et al., 2014), and open urban spaces (Liu et al., 2021; Wang et al., 2022; Zhou et al., 2019) (Table 7).

The concentrations of HMs in the collected samples were determined using Proton-Induced X-ray Emission (PIXE), Wavelength-dispersive X-ray Fluorescence (WDXRF) Spectrometry, and Inductively Coupled Plasma-Mass Spectrometry (ICP-MS).

The highest number of research in this field was published in China (4 articles). According to the results, the concentrations of Cr, As, Pb, and Ni in hospitals (Slezakova et al., 2014) and As in the subway systems were higher than the limit values established by the WHO (Fig. 10).

Hospitals are regarded as a unique and significant category of indoor public places, as the air quality in these establishments considerably influences the future health outcomes. Nevertheless, data on indoor air quality in these settings are scarce, specifically on exposure to HMs. An assessment of the possible risks related to exposure to trace metals in a hospital setting near Portugal's busiest highway and bordered by national roads revealed that potential risks for As and Cr-bound $PM_{2.5}$ were higher than the USEPA guideline by up to 62 and 5 times, respectively, for all age groups (Slezakova et al., 2014).

In China, HM concentrations, such as Cd, Cr, Co, Pb, and Ni, in different functional areas in Hefei and five types of Northeast cities, were higher than the limit values set by the WHO (Zhou et al., 2019). Due to the increase in population, there has been a noticeable increase in the traffic volume on Chinese roads. In 2021, the average traffic volume on national roads (including highways) in China was 14,993 vehicles per day, indicating a slight increase of 4.2 % from the previous year (Lin et al., 2021). Vehicle emissions, such as car oil usage, car component collisions, and wear of tires, brake pads, and engines, were identified as the sources of Cu, Cr, Ni, Pb, and Zn (Sulaiman et al., 2017). Past studies revealed that vehicle emissions contributed mainly to Pb content in indoor dust (Praveena et al., 2015; Tong and Lam, 2000). Lead is persistent in the environment, and once it is released into the

Table 6
The characteristics of studies on HM exposure in office and commercial environments.

Country	Sampling environment	Analysis method	Duration of sampling	Point of sampling	Sample size	References
Office						
Malaysia	Office	ICP-MS	12 months	Malaysia	–	Othman et al., 2016
Malaysia	Office	ICP-MS	12 months	Malaysia	48	Othman et al., 2018
India	Offices	Atomic Absorption Spectrometry	3 months	India	96	Massey et al., 2016
China	Indoor environments including office	ICP-MS	_	China	88	Lu et al., 2019
Commercial						
Iran	Tobacco cafés	AAS	3 months	Tehran	–	Masjedi et al., 2020
China	Commercial restaurants	ED-XRF	3 months	Guangdong	–	Dai et al., 2018
India	Shops, commercial centers	AAS	3 months	Agra	96	Massey et al., 2016
Turkey	Restaurants	ICP-MS	3 months	Kocaeli	14	Taner et al., 2013

- Not defined.

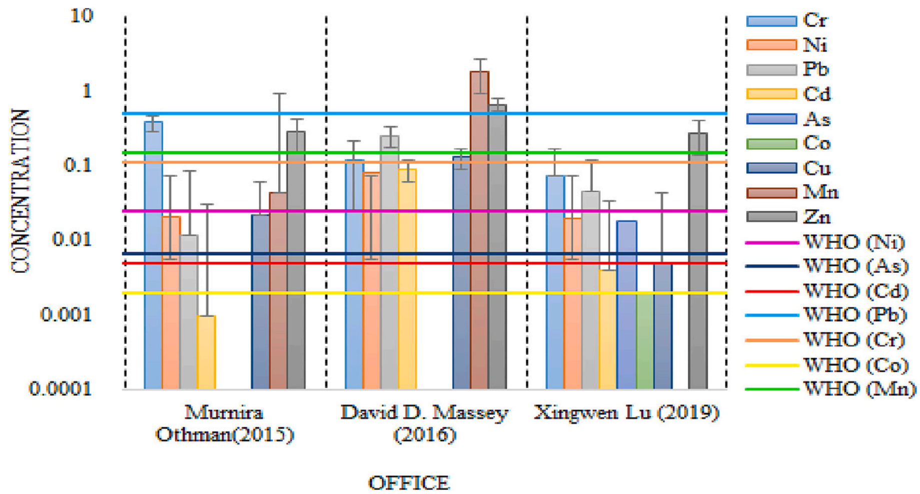

Fig. 8. Comparisons of HM concentrations in office environments (μg m^{-3}).

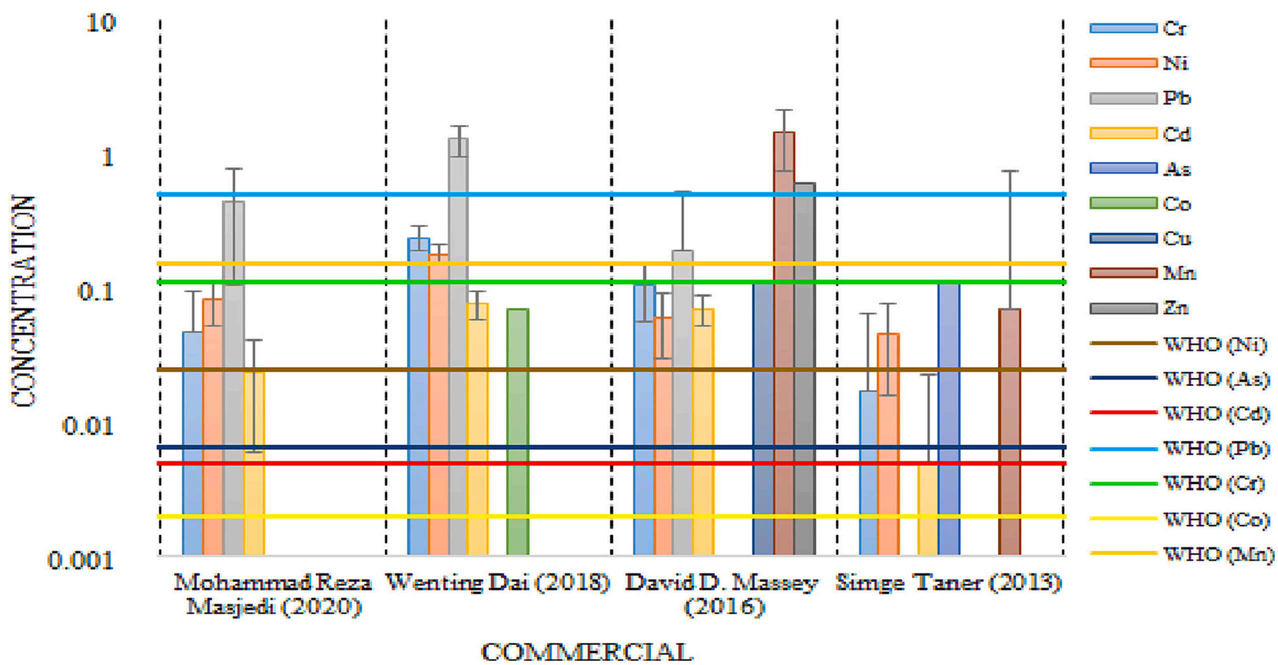

Fig. 9. Comparisons of HM concentrations in commercial environments (μg m^{-3}).

Table 7
The characteristics of studies on HM exposure in other types of environments.

Country	Sampling environment	Analysis method	Duration of sampling	Point of sampling	Sample size	References
China	Subway stations	WDXRF	9 months	Nanjing	182	Ji et al., 2022
China	Different functional zones (the chief district, commercial district, and industrial district)	ICP- OES	3 months	Hefei (Chief District)	–	Zhou et al., 2019
China	Street	ICP-MS	3 months	Provincial central city	255	Liu et al., 2021
China	Rooftop of the party and government building	PCA	12 months	Pingyao	236	Wang et al., 2022
Portugal	Hospital	PIXE	-	Oporto	28	Slezakova et al., 2014

- Not defined.

environment, it remains in the circulation without undergoing natural degradation. Despite the prohibition of Pb in gasoline, trace levels of Pb still remain at a level of up to 13 mg L^{-1}. The intensive use of gasoline, particularly in cities with heavy traffic, can lead to Pb air pollution. Studies of Zhao et al. (2006), Kong et al. (2011), and Zhong et al. (2014) determined that high concentrations of Zn originated from tire wear, galvanized materials, indoor products, and rubber production. Contamination with Cd was considered in the investigated cities,

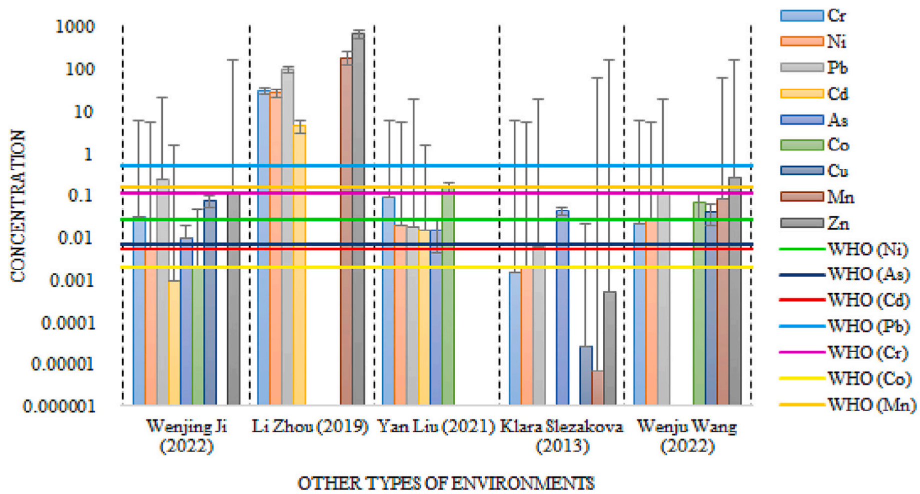

Fig. 10. Comparisons of HM concentrations in other types of environments (μg m^{-3}).

indicating that it was the main concern in the urban air of China. Central regional and provincial cities, as well as resource-based cities, were found to have the highest levels of Cd contamination of urban air (Liu et al., 2021). These findings were consistent with the study by Chen et al. (2015), who also indicated that Cd was the element causing the worst urban air quality in China. However, the sources of Cd contamination varied from city to city for the investigated cities. For example, Cd hotspots with high concentrations were found in the western part of the Shenyang Tiexi industrial district. In 2000, the authorities closed the Shenyang smelting plant after 64 years of operation due to significant air pollution (Sun et al., 2010). These results indicate that large-scale smelting activities have a major impact on urban air pollution in the region. In the study of Chen et al. (2014), a Cd contaminated hotspot was revealed between the second and third ring roads in the western section of the downtown area of the city of Xi'an in China and Cd was mainly derived from industrial sources, traffic, and urban waste. In contrast to the trends observed in Shenyang and Xi'an, a significant Cd concentration was detected in Changsha along the Xiangjiang River, upstream of which large-scale Pb, Zn, and Cd mining was carried out. Therefore, Cd discharges grown considerably because of the increasing anthropogenic input brought about by China's rapid urbanization, industrialization, and economic growth (particularly since the late 1970s). In approximately 50 % of the cities under investigation, moderate levels of pollution with Pb, Cu, Zn, and Hg were indicated. The decreasing levels of atmospheric pollution with HMs in urban areas were as follows: industrial-based cities > more developed cities > metropoles > undeveloped cities. Han et al. (2017) indicated that machinery industries were the primary source of Co, while human activities such as steel plants, metallurgy industries, and commercial districts were responsible for the presence of Zn and As. Iron is an element derived from the soil as a result of the high natural abundance in the Earth's crust.

3.6. Health risk assessment

3.6.1. Non-carcinogenic risk

In this part studies on non-carcinogenic health effects of HMs present in ambient particulate matter through inhalation exposure were investigated (Table 8).

Despite the higher levels of HM in the studies of Al-Harbi et al. (2021), Li et al. (2022a), Lu et al. (2019), Li et al. (2016), Tashakor et al. (2022), Rabha et al. (2018), Lin et al. (2015), and Hashemi et al. (2020), in these environments the lowest values of non-carcinogenic risk reported (HQ < 1). However, Albar et al. (2020) reached different results.

In household environments, Mn contributed the most to the non-carcinogenic risk value, followed by Cd, Pb, As, Ni, Cr, Co, and Cu. In addition, the non-carcinogenic risk values were reported in 4 studies, of which Albar et al. (2020) reported HQ values exceeded the acceptable risk, with HI values of 3.19 for Ni, 1.60 for Cu, and 95.3 for Mn, AlMulla et al. (2022) with HQ values of 35.58 for Cd and 1.57 for Cr, Rohra et al. (2018) with HQ values of 2.89, and Li et al. (2022a) with HQ values of 2.09 for Cd, 26.53 for Pb, and 27.86 for As. According to the HQ values, there were no adverse effects in people (adults or children) due to exposure to Zn in all the investigated environments.

According to the examined literature on household environments, children were revealed to be more susceptible than adults to the non-carcinogenic effects of exposure to HMs due to their unique physiological makeup, such as participation in play activities and higher respiratory rates relative to their body weight (Zhao et al., 2014). Agricultural, industrial, and household areas in Saudi Arabia showed a considerable health risk to Cd, Cr, and Pb, and the health risks for children due to exposure to indoor dust were higher than for adults. Among the sites, the highest HQ for these metals was found for the sites of the Al-Qatif industrial area, and among the metals Cd was the highest (AlMulla et al., 2022). In cities of northern China, the non-carcinogenic risk values for both heating and non-heating periods in household environments were higher than the acceptable level, indicating that in these areas people can be exposed to non-carcinogenic risks by exposure to HMs in urban air (Li et al., 2022a). These results indicated that ambient exposure to multiple metals posed a greater non-carcinogenic health risk than exposure to these metals separately.

In educational environments, non-carcinogenic risk values did not exceed the acceptable level (HQ = 1) for both adults and children, as indicated by the calculated HQ values for Pb, Cd, Cr, As, Ni, Cu, and Zn. All HMs investigated in the office environments had HQ values below the acceptable level, indicating negligible non-carcinogenic risk from inhalation exposure from HMs in air dust. Non-carcinogenic risk values in office environments were below the acceptable level, indicating an acceptable risk in these places. In the section related to commercial environments, the non-carcinogenic risk values of Cr, Pb, and Cd in cafes studied in Tehran, Iran (Masjedi et al., 2020) and As in barbeque restaurants in Turkey (Taner et al., 2013) were higher than the acceptable level.

In studies conducted in other types of environments, the non-carcinogenic risk value of As in the hospital (Slezakova et al., 2014) and Co, Ni, and Zn in the subways (Ji et al., 2022) was higher than the acceptable level (HQ > 1). Sources derived from platforms, concourses, and office spaces in the metro station accounted for 54 %, 81 %, and 71 % of non-carcinogenic risks.

We found that the average HQ values of Zn, Pb, Cu, Cr, Co, Ni, As, and Cd in educational and administrative environments were lower than

Table 8
Non-carcinogenic risk (HQ) values of HMs in the different environments of the systematic review.

Country	Health risks	Non-carcinogenic risk (HQ)									References
		Cd	Pb	As	Co	Cr	Ni	Zn	Cu	Mn	
Households											
Saudi Arabia	HI	2.83E-01	-	-	1.05E-01	1.02E-01	3.19	6.68E-01	1.60	9.53E+01	Albar et al., 2020
Saudi Arabia	HI	35.58	0.552	-	-	1.5781	-	1.78E-02	1.054E-01	2.49E-02	AlMulla et al., 2022
China	HQ	8.22E-02	-	1.60E-01	-	-	2.45E-02	-	-	1.32E-01	Wu et al., 2019
China	HQ	1.71E-01	1.57E-01	1.94E-01	1.97E-01	8.80E-02	5.41E-01	1.35E-01	8.63E-02	5.76E-02	Li et al., 2016
India	HQ	-	-	-	2.89	4.75E-01	-	-	-	2.68E-01	Rohra et al., 2018
Kuwait	HI	-	1.69E-01	8.55E-02	3.53E-03	4.08E-01	5.98E-02	7.94E-03	1.23E-02	5.05E-03	Al-Harbi et al., 2021
China	HI	1.50E-03	1.50E-01	1.74E-01	-	-	-	-	6.81E-03	1.35E-02	Yang et al., 2015
China	HI	-	6.6 E-01	8.38E-03	1.10E-03	2.26E-02	2.12E-03	9.42E-03	1.76E-02	-	Lin et al., 2015
India	HQ	5.00E-04	-	-	2.02E-02	-	8.30E-03	-	-	9.70E-03	Rabha et al., 2018
China	HQ	1.45E-02	3.13E-02	3.97E-02	1.01E-01	2.61E-03	-	-	-	2.73E-02	Li et al., 2021
China	HQ	3.13E-02	3.37E-01	5.87E-01	-	3.92E-01	1.63E-02	4.83E-03	1.21E-02	1.25E-02	Li et al., 2022a
Iran	HI	7 E-03	2.25 E-02	1.7 E-01	2.15 E-02	1.125 E-01	4.25 E-03	-	-	-	Tashakor et al., 2022
China	HQ	2.095	26.53	27.86	-	-	-	-	-	-	Li et al., 2022b
South Africa	HQ	1.44E-05	-	9.96E-10	3.33E-03	2.15E-06	-	1.03E-03	1.67E-03	-	Lion and Olowoyo, 2021
Iran	HQ	2.5E-06	1.3E-04	-	-	4E-04	7E-08	7.25E-06	3.32E-06	-	Hashemi et al., 2020
School											
Malaysia	HQ	5.13E-05	7.63E-06	3.15E-04	8.40E-05	8.40E-05	4.16E-04	-	1.38E-04	-	Tan et al., 2018
Malaysia	HQ	1.67E-05	6.70E-04	-	3.58E-04	8.88E-01	5.75E-02	-	4.10E-05	-	Mohamad et al., 2016
Saudi Arabia	HI	1.50E-02	2.12E-01	1.63E-01	1.09E-02	9.13E-02	2.73E-03	7.03E-03	1.32E-02	5.77E-02	Alghamdi et al., 2019
Poland	HQ	8.80E-03	-	3.95E-02	1.56E-02	2.87E-02	-	-	-	1.66E-01	Mainka, 2021
Nigeria	HQ	1.36E-09	-	1.56E-10	-	4.84E-11	6.26E-08	-	-	4.25E-09	Famuyiwa and Entwistle, 2021
Nigeria	HQ	1.27E-08	-	-	-	9.97E-07	2.86E-07	-	-	-	Famuyiwa et al., 2023
Iran	HQ	7.60E-07	5.17E-05	-	3.50E-07	2.14E-07	2.89E-08	2.05E-06	4.33E-07	-	Hashemi et al., 2020

Malaysia	HQ	2.50E-07	-	1.12E-05	1.57E-07	3.30E-07	-	5.05E-07	1.15E-07	1.99E-07	Othman et al., 2019
Malaysia	HQ	1.44E-03	-	-	6.01E-02	1.92E-04	6.25E-03	-	-	9.31E-03	Othman et al., 2022
Malaysia	HQ	2.41E-01	1.59E-01	5.78E-01	5.91E-02	3.71E-02	5.64E-01	-	-	6.61E-01	Alias et al., 2021
University											
Malaysia	HQ	-	1.02E-06	3.65E-05	-	-	-	-	-	-	Sulaiman et al., 2017
Iran	HI	6.78E-05	278E-04	-	-	1.0244E-03	-	-	1.969E-05	-	Vosoughi et al., 2021
China	HQ	5.00E-03	5.00E-03	8.75E-03	4.00E-03	3.55E-02	9.75E-03	4.00E-03	5.00E-03	-	Lu et al., 2019
China	HI	2.82E-01	3.97E-01	4.68E-01	2.3E-01	2.2 E-01	-	-	7.5E-02	6.53E-01	Bao et al., 2019
China	HQ	3.77E-05	1.47E-04	1.34E-04	6.81E-06	8.48E-03	-	5.54E-06	5.00E-06	2.10E-02	Wang et al., 2019
Turkey	HI	5.70E-03	2.20E-02	1.00E-01	6.40E-03	4.10E-02	3.80E-02	-	-	-	Arı, 2020
Office											
Malaysia	HQ	1.15E-01	-	-	4.16E-01	1.72E-01	-	-	-	2.52E-01	Othman et al., 2016
Malaysia	HQ	2.80E-09	-	9.96E-10	3.12E-09	2.23E-08	-	-	-	5.93E-10	Othman et al., 2018
Commercial											
Iran	HI	1.06	2.3	-	1.36	2.84E-03	-	-	-	-	Masjedi et al., 2020
Turkey	HQ	1.14E-01	-	1.69	7.50E-01	3.88E-02	-	-	-	3.15E-01	Taner et al., 2013
Other types of environments											
China	HI	-	-	2.14E-01	2.27	7.3E-02	8.91	1.38	-	-	Ji et al., 2022
China	HQ	-	9.53E-01	-	9.12E-01	9.16E-01	6.53E-01	9.51E-01	9.45E-01	5.54E-01	Wang et al., 2022
China	HQ	2.08E-03	-	3.38E-03	5.80E-04	2.74E-03	8.30E-02	-	-	-	Liu et al., 2021
China	HI	7.06E-04	-	-	-	1.27E-03	6.79E-04	2.48E-03	-	8.31E-03	Zhou et al., 2019
Portugal	HQ	-	-	4.43	5.00E-02	2.54E-03	-	-	-	4.52E-02	Slezakova et al., 2014

[1]HQ values were estimated using the RfD_{inh} toxicity parameter.

Non-carcinogenic risk (HQ)	
	acceptable risk
	significant risk

-Not defined.

the acceptable level compared to those in other environments investigated. Additionally, it should be noted that the average non-carcinogenic risk values associated with inhalation of HMs in commercial environments were significantly higher than those observed in urban outdoor environments (Liu et al., 2021; Wang et al., 2022; Zhou et al., 2019). Due to the non-carcinogenic effects of Pb, Cd, Cr, As, Co, and Ni, the non-carcinogenic risk values associated with exposure to HMs in household, commercial, and service environments such as hospitals and metros, should attract the attention of decision-makers.

The difference in the non-carcinogenic risk due to inhalation exposure to HMs from air dust in the office and educational environments compared to other environments can be due to several factors, including different environmental conditions, ventilation systems, specific activities of employees in each sector, the absorption rate of HMs, sampling technique, and volume of air measured. Based on these results, some environments were stated to pose higher health risks, which should be examined more closely. In addition, the variations in the HM concentrations between administrative and educational buildings compared to other indoor environments can vary depending on the density of the materials and the types of equipment used in each environment. The most significant way to prevent the accumulation of HMs in interior spaces is to increase the air exchange rate. In general, the non-carcinogenic risks associated with As, Cd, Co, and Cr in indoor dust should receive more attention.

3.6.2. *Carcinogenic risk*

The carcinogenic risk for Pb, Cd, Cr, Co, Ni and As was evaluated, as there are no SF values for Cu, Zn, and Mn (Hu et al., 2012). Table 9 shows the CR values of exposure to HMs in the ambient air by inhalation pathway.

The carcinogenic risk values of HM from household environments reported in the studies of Li et al. (2022a), Li et al. (2016), Lu et al.

Table 9
Carcinogenic risk (CR) values of HMs in the different environments of the systematic review.

Country	Carcinogenic Risk (CR)						References
	Cd	Pb	As	Co	Cr	Ni	
	Household						
Saudi Arabia	2.00E-07	-	-	3.30E-06	1.82E-05	1.23E-05	Albar et al., 2020
Saudi Arabia	8.19E-08	1.50E-02	-	-	2.83E-07	-	AlMulla et al., 2022
China	3.16E-07	1.40E-08	2.22E-06	-		1.04E-07	Wu et al., 2019
China	-	4.16E-06	1.59E-04	-	6.48E-05	-	Li et al., 2016
India	-	1.55E-06	-	6.43E-06	7.40E-04	-	Rohra et al., 2018
Kuwait	-	1.85E-06	1.67E-04	6.49E-05	5.21E-03	-	Al-Harbi et al., 2021
China	1.95E-10	-	2.82E-09	-	-	-	Yang et al., 2015
China	-	-	3.85E-09	1.87E-09	2.73E-07	5.77E-09	Lin et al., 2015
India	3.08E-06	-	-	2.30E-06	-	-	Rabha et al., 2018
China	1.97E-07	2.28E-08	1.94E-06	2.57E-07	2.38E-06	-	Li et al., 2021
China	1.43E-04	-	8.88E-05	1.41E-04	8.88E-05	1.11E-05	Lu et al., 2019
China	4.74E-06	-	3.41E-04	-	2.30E-04	3.13E-05	Li et al., 2022a
Iran	1.92E-09	7.51E-07	7.21E-05	5.87E-09	4.12E-05	-	Tashakor et al., 2022
China	1.53E-06	1.55E-06	6.13E-06	-	-	-	Li et al., 2022b
South Africa	1.83E-09	-	4.04E-09	-	3.52E-08	-	Lion and Olowoyo, 2021
Iran	-	-	-	-	4.15E-09	-	Hashemi et al., 2020
School							
Malaysia	1.30E-07	9.11E-11	5.69E-08	5.60E-08	-	-	Tan et al., 2018
Saudi Arabia	8.13E-05	6.17E-06	7.40E-06	1.83E-04	1.01E-04	4.90E-04	Alghamdi et al., 2019
Poland	8.80E-08	-	2.48E-06	7.28E-08	2.44E-04	-	Mainka, 2021
Nigeria	-	1.37E-07	8.19E-07	-	1.42E-06	1.82E-06	Famuyiwa and Entwistle, 2021

Nigeria	2.73E-11	1.00E-06	-	-	2.56E-05	1.80E-11	Famuyiwa et al., 2023
Iran	-	-	-	-	2.93E-08	-	Hashemi et al., 2020
Malaysia	-	3.94E-09	6.26E-10	2.80E-10	5.53E-10	-	Othman et al., 2019
Malaysia	2.58E-08	8.16E-09	-	1.68E-07	4.84E-05	-	Othman et al., 2022
Malaysia	1.29E-06	1.23E-08	3.20E-06	2.26E-06	2.67E-05	7.81E-08	Alias et al., 2021
University							
Iran	1.93E-12	-	-	-	6.69E-11	-	Vosoughi et al., 2021
China	1.16E-04	-	8.52E-05	5.91E-05	8.05E-06	-	Bao et al., 2019
China	1.10E-07	-	8.34E-08	6.53E-09	5.83E-07	-	Wang et al., 2019
China	1.41E-04	-	8.88E-05	1.41E-04	1.11E-05	1.11E-05	Lu et al., 2019
Turkey	9.20E-08	-	1.30E-06	2.30E-07	3.60E-06	1.90E-06	Arı, 2020
Office							
Malaysia	1.38E-07	1.35E-03	-	4.84E-07	7.17E-05	-	Othman et al., 2016
China	1.41E-04	-	8.88E-05	1.41E-04	1.11E-05	1.11E-05	Lu et al., 2019
Malaysia	-	7.12E-10	-	1.98E-11	3.72E-11	-	Othman et al., 2018
India	1.6E-07	-	-	-	1.4E-07	1.9E-07	Massey et al., 2016
Commercial							
Iran	-	-	-	-	2.06E-03	-	Masjedi et al., 2020
China	2.30E-05	-	-	-	-	-	Dai et al., 2018
India	1.20E-07	-	-	-	1.25E-07	1.40E-07	Massey et al., 2016
Turkey	7.34E-07	4.21E-08	3.89E-05	9.00E-07	1.16E-04	-	Taner et al., 2013
Other types of environments							
China	-	2.66E-05	-	6.85E-05	7.41E-07	1.37E-06	Wang et al., 2022
China	3.12E-08	2.19E-10	9.36E-08	5.73E-09	2.30E-05	1.92E-06	Liu et al., 2021
China	1.04E-09	2.04E-07	-	-	1.91E-06	-	Zhou et al., 2019
China	-	4E-06	3.29E-05	5.257E-05	2.887E-05	4.2E-05	Ji et al., 2022
Portugal	-	2.48E-08	6.31E-05	2.64E-07	4.65E-06	-	Slezakova et al., 2014

Carcinogenic Risk (CR)	**Negligible risk**	**Possible risk**	**Probable risk**	**Definite risk**

-Not defined.

(2019), Al-Harbi et al. (2021), Rohra et al. (2018), and AlMulla et al. (2022) were within the definite risk (CR $>1 \times 10^{-4}$) according to the USEPA methodology. This indicated that residents of the cities investigated, especially children, were potentially exposed to a severe carcinogenic risk. Only in 3 studies reviewed in household environments the carcinogenic risk values were at the acceptable level (CR $<1 \times 10^{-6}$) (Lin et al., 2015; Lion and Olowoyo, 2021; Yang et al., 2015).

Children are more exposed than adults to the carcinogenic effects of HMs, according to the results of the carcinogenic risk assessment. For example, Al-Harbi et al. (2022) revealed that the mean carcinogenic risk value for adults for Cr was equal to 4.85×10^{-3} and for As, 2.51×10^{-3}. For children, the mean CR value for Cr was equal to 5.57×10^{-3} and for As, 2.89×10^{-3}. Investigations revealed the following decreasing order of carcinogenic risk values in household environments: Co > Cd > Pb > As > Cr. The CR values for Ni were at the probable risk level for these elements in household environments.

Epidemiological studies revealed a link between exposure to Pb in the environment and an increased risk of cancer. Through oxidative damage, interactions with DNA binding proteins, and inhibition of the DNA repair system, Pb can cause carcinogenicity (Silbergeld, 2003). Epidemiological investigations demonstrated a correlation between the presence of As and the elevated incidence of lung, bladder, and skin cancers (Rossman, 2003). There is substantial evidence that Cd causes cancer, as the mechanisms involved in Cd carcinogenesis encompass suppression of gene expression, induction of oxidative stress, and suppression of gene expression (Bishak et al., 2015). Values of CR and HQ for HMs in heating periods (it means the time when it is cold outside and combustion of fuels is needed to increase the temperature inside), are more than non-heating periods (Li et al., 2022a; Li et al., 2022b). In Changchun city, the carcinogenic risk of Ni for adults in the heating period was equal to 2.11×10^{-5} and in the non-heating period 2.17×10^{-5} and for children in the heating period was equal to 14.06×10^{-5} and in the non-heating period 4.19×10^{-5} (Li et al., 2022a). Also, based on the results of studies of Lion and Olowoyo (2021), Othman et al. (2022), Othman et al. (2018), and Wang et al. (2019), the CR and HQ values for indoor HMs were higher than those of outdoor HMs.

In educational environments, the carcinogenic risk values from the studies of Alghamdi et al. (2019), Mainka (2021), Bao et al. (2019), and Lu et al. (2019) were higher than the acceptable level. Metals with definite carcinogenic risk values (1×10^{-4}) in educational environments include Cd, Cr, Co, and Ni. The CR values of HMs in school environments decreased in the following order: Cr < Ni < Co < Cd < As < Pb. In contrast, the carcinogenic risk values for Pb and As in school environments were at a possible level for children. The carcinogenic risk values from the studies of Famuyiwa and Entwistle (2021), Alias et al. (2021), and Othman et al. (2022) indicated probable risks. Zheng et al. (2013) assessed the effects of exposure to Cr, Ni, and Mn on lung function in 144 schoolchildren, aged from 8 to 13 years. Their studies revealed that boys aged from 8 to 9 years had a lower forced vital capacity (FVC) of lung function (1859 vs. 2121 ml, $p = 0.003$). The contribution of the particular HMs to the carcinogenic risk in university environments decreased in the following order: Cd > Co > As > Cr > Ni. The carcinogenic risk values of As, Cd, Co, Cr, and Ni reported in the study of Lu et al. (2019) were within a range of 1.11×10^{-5} to 1.41×10^{-4}, indicating that the acceptable risk level of 1×10^{-6} was exceeded. In the reviewed articles, all reported HM values indicated the lowest carcinogenic risk in the summer season and the highest in the winter season (Bao et al., 2019; Mainka, 2021).

Among the HMs evaluated in office environments, Pb caused the highest carcinogenic risk value for adults according to the study of Othman et al. (2016) with the CR value of 1.35×10^{-3}. The total CR values found in the studies of Lu et al. (2019) for HMs ranged from 1.11 $\times 10^{-5}$ to 1.41×10^{-4}. The reason explaining this situation could be overcrowding, carrying out various laboratory activities, and the proximity to the main traffic routes of the city. Arı (2020) estimated that the carcinogenic risk of Cr, Ni, and As exposure was within the possible risk range in an office room at the university.

The CR values for HMs in commercial environments exceeded an acceptable level of 1×10^{-6}, indicating potential carcinogenic risks of exposure to multiple metal inhalation. The carcinogenic risk values of HMs in the study of Massey et al. (2016) were lower than 1.00×10^{-6}, which revealed a negligible risk. Among the HMs evaluated in commercial environments, the highest carcinogenic risk values were stated for Cr in cafes (CR = 2.06×10^{-3}) (Masjedi et al., 2020) and barbeque restaurants (CR = 1.16×10^{-4}) (Taner et al., 2013). The CR values for Cr in commercial environments also reflect the potential for carcinogenic effect occurrence. The CR values for Cd in commercial restaurants (CR = 2.30×10^{-5}) (Dai et al., 2018), and As (CR = 3.89×10^{-5}) in barbeque restaurants (Taner et al., 2013) were in the probable risk level. IARC has designated Cr(VI) as a Group 1 carcinogen. Cr(VI) is readily absorbed from the respiratory system and enters the cells, where various reductants, including glutathione and ascorbate, can reduce it. This reduction produces glutathione radicals, Cr(V) and Cr(IV), which can target DNA and cause Cr(VI)-induced carcinogenesis (Ding and Shi, 2002).

Among studies related to urban, industrial, subway, and hospital environments, the average values of the carcinogenic risk for all environments were within the range of probable carcinogenic risks. Furthermore, the risks associated with As, Co, Cr, and Ni contents on the subway platforms in China exceeded the acceptable risk level of 1×10^{-5}, indicating that city commuters were potentially exposed to severe carcinogenic risk. Slezakova et al. (2014) found that in therapeutic settings, such as hospitals, the total carcinogenic risks of HMs exceeded the recommended level highly (up to 67 times) for all age groups, clearly indicating that exposure to HMs in fine particles poses a significant risk. These results revealed alarming HM concentrations in therapeutic settings and the possibility of related serious side effects in patients.

When the human body is exposed to HMs from air inhalation, non-dietary ingestion, such as soil/dust, or contact with the skin, it accumulates these HMs inside. HMs exhibit a propensity for bioaccumulation within various human tissues, namely placenta, umbilical cord blood, blood and serum, hair, and urine. HMs have the potential to cause a variety of diseases, such as cancer, abnormalities in mental health, neurodevelopment, thyroid dysfunction, and overall decline in physical health. These harmful effects are primarily due to DNA damage and changes in gene expression (Mitra et al., 2022). For example, the toxic effects of excessive Ni concentrations include liver, kidney, and brain damage. IARC classified Ni as a group 1 carcinogen. Ni can induce DNA damage by binding directly to DNA, inhibiting the damage repair system, and stimulating reactive oxygen species (ROS) production (Guo et al., 2019). It is well-recognized that prolonged exposure to HMs can have adverse health effects. Health risk values in most of the investigated cities were within the range of possible to probable risk. The risk value is also expected to increase as traffic levels and population density increase. The primary sources of HMs in urban environments include industrial chemical emissions, coal combustion, dust deposition, automobile emissions, and suspended atmospheric materials. According to studies, environmental protection authorities should pay closer attention to the health concerns generated by HMs in urban environments. Since indoor dust is the main source of HMs, further studies are needed on indoor dust exposure, particularly in children.

The study revealed that indoor environments, particularly educational settings, can expose children and students to high levels of HMs. Exposure to HMs can have detrimental impacts on the health of children and students. Therefore, it is crucial to closely observe the health conditions of children and students in areas close to industrial sites to guarantee their protection against prolonged exposure to HMs, which can lead to permanent health consequences. One method to enhance air quality in educational facilities is accurately assessing the supply based on the demand. Increasing officials' awareness, especially managers', is fundamental to improving air quality in educational environments. It is essential to consistently utilize the indoor air quality index to continuously monitor the air quality and then evaluate the level of health risk for children and students in educational facilities. However, proposing strategies to enhance air quality in educational buildings can substantially affect indoor air quality. For example, air purifiers can lower acceptable particle concentrations by 40 %. Therefore, better studies and monitor these environments are needed to provide appropriate control techniques and recommendations for this public health issue.

There are currently guidelines for the materials and products used in indoor residential and public spaces. However, it is essential to note that these fixtures and fittings can remain in place for many years, but they may not be compliant with today's standards. Consequently, this indicates that older interior spaces that are poorly constructed or maintained and those that contain outdated furniture or personal belongings may have higher risk values. Air purifiers can protect human health from heavy metals by absorbing particle matter in the interior air. As a result, when pollution sources are challenging to manage, we may employ air purifiers to produce a more pleasant and healthful indoor environment. It is essential to consider the impact of building design and production on pollution emissions, especially in public spaces, as this is relevant to public health. Utilizing more efficient airflow and pollutant transport modeling techniques in enclosed spaces can help elucidate variations in risk ratings among residential structures located in different locales or at varying distances from pollution sources like roadways. There is a growing focus on designing sustainable buildings and implementing low-energy and automated heating/cooling systems in buildings. These advancements can create healthier and more environmentally friendly indoor environments. In conclusion, the study emphasizes the critical nature of future deliberations regarding environmental HM monitoring, specifically legislation addressing this issue via control measures applicable to indoor and outdoor environments.

4. Conclusions

Environmental pollution with heavy metals (HMs) is one of the biggest concerns worldwide as environmental exposure to heavy metals has grown as a result of the growing usage of chemical products, and IARC has proven that some of them are carcinogenic and mutagenic. As local governments, industries, and businesses may lack measures to mitigate heavy metal emissions, this systematic review investigated inhalation exposure to heavy metals in environmental situations and related health risks worldwide using data from five credible databases. In this study, the levels of pollution and related health risks posed by Pb, Cd, Cr, Cu, Zn, Ni, As, and Co in ambient air were analyzed. Based on the results obtained, it was revealed that HM concentrations in most household environments were higher than the WHO guideline values, reflecting moderate levels of pollution and the strong possibility of being influenced by the anthropogenic activities of HMs. Most of the school, university, and office environments studied were characterized by low to moderate levels of air pollution. In contrast, in commercial environments investigated levels of HM pollution were high. The HQ values indicated that in some household environments, cafes, hospitals, restaurants, and subways the non-carcinogenic risk was significant due to inhalation exposure to HMs in humans. Additionally, the carcinogenic risk stated in household environments for Cd, Pb, Cr, As, and Co, in educational environments for Cd, Cr, Ni, and Co, in office environments for Pb, Cd, and Co, and in commercial environments for Cr exceeded the acceptable level (1×10^{-4}). These findings provide valuable information on the distribution of HMs in various indoor and outdoor environments and highlight the environments in which the highest attention should be paid for implementing effective interventions to reduce HM exposure and related health risks in the future. Considering the biological conditions of children and their higher vulnerability during exposure to inhaled pollutants, continuous monitoring of air quality in the educational and household environments is suggested.

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Amir Hossein Khoshakhlagh: Writing – review & editing, Writing – original draft, Supervision, Software, Resources, Project administration, Methodology, Investigation, Data curation, Conceptualization. **Safiye Ghobakhloo:** Writing – original draft, Software, Project administration, Investigation, Formal analysis, Conceptualization. **Willie J.G.M. Peijnenburg:** Writing – review & editing. **Agnieszka Gruszecka-Kosowska:** Writing – review & editing. **Domenico Cicchella:** Writing – review & 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.

Data availability

Data will be made available on request.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2024.172556.

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