



Deciphering the origin of Cu, Pb and Zn contamination in school dust and soil of Dhaka, a megacity in Bangladesh

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Abstract

In recent decades, anthropogenic activities have resulted in road dust and roadside soil hosted metal(loid)s pollution in the urban environment. In the South-Asian megacity “Dhaka”, schools are situated in the areas with high population density and high traffic emissions. As the school-going children are the most vulnerable receptor, school premises in Dhaka city represent an important yet overlooked exposure point to contaminated dust and soil. Therefore, the present study investigated the metal(loid)s (Cu, Pb, Zn and As) pollution in dust and soil at school compounds, explored their possible sources and estimated the associated human health risk. This study revealed that dust contained higher concentration of metal(loid)s than soil, and the Azimpur Govt. Girls School & College was identified as the most contaminated site. The enrichment of school dust with Cu, Zn and Pb were strictly related to the dense population and substantial traffic activity in the study areas. Arsenic content in school soil was several folds higher than its concentration in the upper crust. Natural and anthropogenic activities possibly posed a synergistic effect on such high soil As. The multivariate statistics suggested that Cu, Zn and Pb were likely to be originated from traffic-related activities, while Zr, Fe, Ti and Rb from natural sources, and K, Sr and Ca from industrial activities. The assessment of health risk suggested the children as a vulnerable receptor and ingestion was identified as the dominant pathway of dust and soil exposure. The hazard index (*HI*) values were lower than unity, suggesting no possible non-cancer health risk. Arsenic posed a lifetime carcinogenic risk to the population in the study area through soil ingestion and dermal adsorption.

Keywords Arsenic · Health risk · Dust · Soil · Metal enrichment

Introduction

In recent decades, high population densities in cities have resulted in extensive human activities, including industrial operations, municipal processes, construction activities and vehicular emissions (Alloway 2013; Amjad et al. 2016; Hong et al. 2018; Han et al. 2020). As a result, various types of toxic organic (such as polycyclic aromatic hydrocarbon) and inorganic substances, particularly heavy metals, have been released into the soil and dust in urban and residential environments (Rahman et al. 2019). The dust is a composite particulate matter originated from diverse sources including soil, deposited construction materials, airborne particulates, soot and fumes discharged from industry and vehicles (Li et al. 2013; Shi et al. 2011; Behrooz et al. 2021). Natural or mechanical forces can transport the fine dust particles into the outdoor and indoor environment such as roadside schools (Moghtaderi et al. 2019). The dust containing metal(loid)s can be easily remobilized and transported into the atmosphere (Ali et al. 2017; Men et al. 2018b), which finally cause long-term

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contamination of air, water and soil (Fujiwara et al. 2011; Liu et al. 2016). In addition, the dust is finer particle than the soils, and it can easily enter into the human body, especially in children in school grounds and classrooms through ingestion, inhalation and dermal contact (Ghanavati et al. 2019a).

The toxic metal(oid)s in dust particles can be bioaccumulated and can affect the circulatory, endocrine and central nervous systems, which lead to growth retardation, kidney disease and various forms of cancer (Thomas et al. 2009; Rahman et al. 2014; Shabbaj et al. 2017; Song and Li 2015). Considering these detrimental effects, school-going children are more vulnerable than adults as they inhale more air per unit of body weight, have less developed respiratory, immune, reproductive, central nervous and digestive systems (Burtscher and Schüepp 2012; Madureira et al. 2016; Salvi 2007; Yu et al. 2018). Besides, they spend a long time in the schools for study in classrooms, playing inside and outside of the classroom as well as in the playground. Such activities may lead to exposure to metal-contaminated dust and soil through inhalation, ingestion and dermal adsorption. Consequently, the health risk assessment model has become an important tool in determining human health risks for metals intake through the aforementioned pathways. This technique has been widely used to assess health risks to the school-going children (Moghtaderi et al. 2019; Rehman et al. 2020) as well as the general population (Kolakkandi et al. 2020; Rahman et al. 2019).

The urban dust and soils act as both source and sink of heavy metals and thus the monitoring of metal content in soil and dust is a potential indicator of contamination in the urban environment (Hou et al. 2019; Men et al. 2018a; Shi et al. 2010). The level of contamination is quantified using several indices, including contamination factor, enrichment factor, geo-accumulation index and pollution load index (Rahman et al. 2019; Saha et al. 2016). Such indices are useful in distinguishing natural and anthropogenic sources of contamination (Barbieri 2016). Besides, multivariate statistics have been employed to identify the possible sources of metal(oid)s in dust and soil in many previous studies (Jin et al. 2019; Othman et al. 2019; Rehman et al. 2020). Recently, several studies have been conducted worldwide on dust heavy metal pollution in the urban environment, such as Kolkata, India (Kolakkandi et al. 2020); Abadan city, Iran (Ghanavati et al. 2019b); Guangzhou, China (Liang et al. 2019); Lahore, Pakistan (Rehman et al. 2020); Petra, Jordan (Alsbou and Al-Khashman 2018); Sydney, Australia (Doyi et al. 2019); Bursa, Turkey (Yaylali-Abanuz 2019) and Philadelphia, USA (O'Shea et al. 2020). In Bangladesh, the study regarding heavy metals in soil and dust in urban areas are scarce. Recently, Rahman et al. (2019) investigated the human health risk for heavy metals in road dust in the capital city of Bangladesh. Ecological risks of heavy metal in different soils of a northwestern city, namely Bogra in Bangladesh, were

studied by Islam et al. (2015). Another study estimated the concentration of major, trace and rare earth elements in street dust in Dhaka city without assessing their pollution and health risks (Ahmed et al. 2007). These limited numbers of studies have provided a glance of heavy metal pollution in street dust and soil in the megacity Dhaka, but none of them considered heavy metal pollution in school dust and soil as well as the associated health risk to the school-going children.

Therefore, this study was aimed to (i) assess the spatial variability of metal(oid) concentrations in the dust as well as soil at different school compounds in Dhaka city, Bangladesh; (ii) evaluate the pollution status using several indices including enrichment factor, geo-accumulation index, contamination factor and pollution load index; (iii) identify the possible sources of pollution through multivariate statistical analysis and (iv) assess carcinogenic and non-carcinogenic health risk to the school-going children.

Materials and method

Study area

The soil and dust samples were collected from the school premises in Dhaka city, the capital of Bangladesh (Fig. 1). Dhaka is located at the centre of Bangladesh, having an area of 1500 km² and a population of ~20 million with a rate of increase of about 7% per year (Saha et al. 2020). The number of vehicles and industries are also multiplying rapidly with the growth in people, and the recent observation of smog in Dhaka city area highlighted air pollution (BRTA 2016). Vehicles, especially two-stroke auto-rickshaws (now banned in Dhaka City) and aged trucks and mini-buses are important factors for most of the air pollution (Chowdhury et al. 2000). Besides, various types of heavy and light industries like textile, glass, ceramic, battery, pharmaceutical, metallurgical and leather processing have been established around Dhaka city, Bangladesh (Rahman et al. 2019).

Bangladesh has a subtropical monsoon climate, which is reflected by marked seasonal variations in temperature, humidity and precipitation throughout the year. Considering the monsoon and metrological condition in Bangladesh, the year is divided into four seasons: (i) pre-monsoon, (ii) monsoon, (iii) post-monsoon and (iv) winter (Rahman et al. 2021a). During pre-monsoon, wind speed becomes moderately strong and the relative humidity observes to increase due to prevailing southwesterly (marine) winds. During the monsoon, the air mass is purely marine in nature and the southwest winds bring plenty of rainfalls (70 to 85% of the annual total). During the post-monsoon, the rainfall, relative humidity and wind speed gradually declines. The wind direction starts shifting back to northeasterly. The winter season (December to February) is dry and accounted for less than 4% of the total

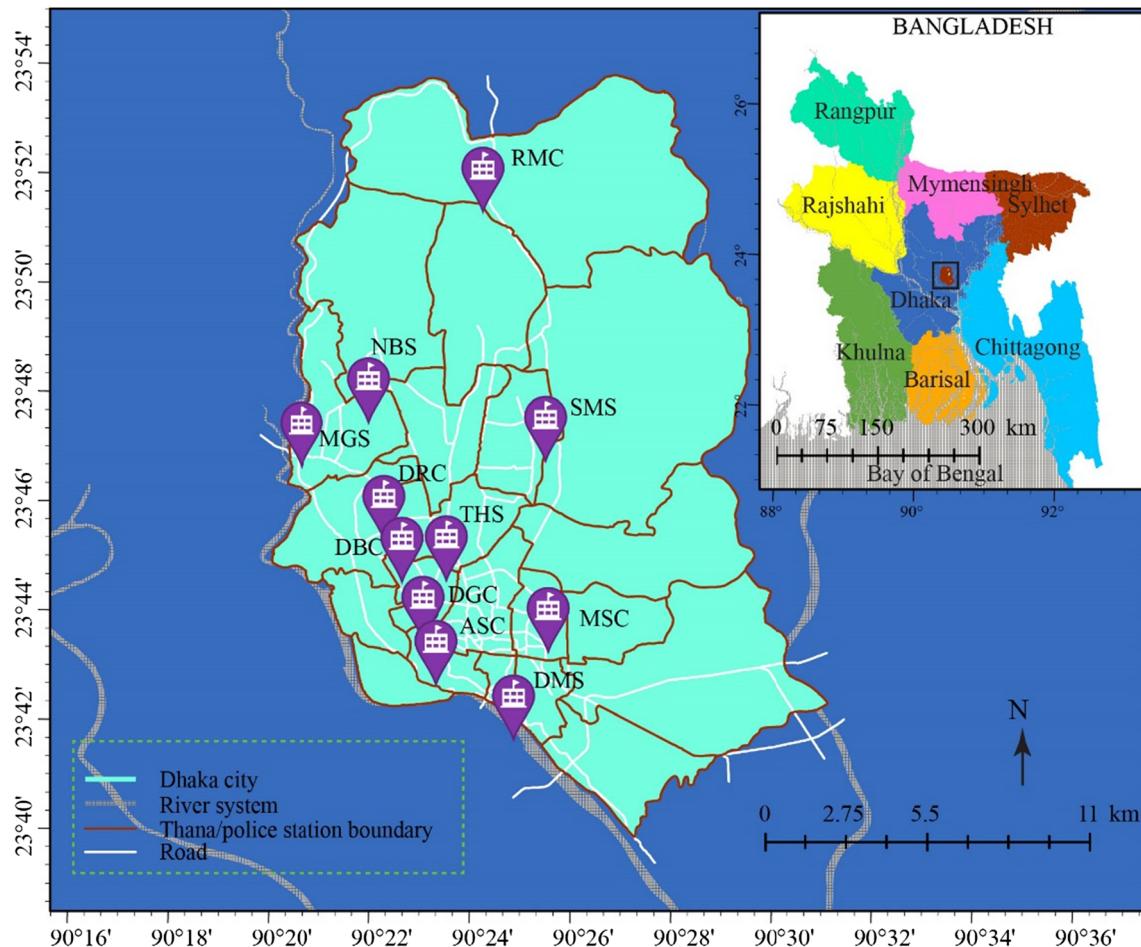


Fig. 1 Study area showing the dust and soil sampling points (red circles) at different schools in Dhaka city, Bangladesh (inset). See Supplementary Table S1 for sampling details

annual rainfall in Bangladesh. Therefore, this season makes the soil dry and changes the weather into low relative humidity. During winter, a centre of high pressure lies over the northwestern part of India (Himalayan zone) and from this high pressure, air enters into Bangladesh from its northwest corner (Rahman et al. 2021a). The meteorological data used in this study was obtained from a local meteorological station in Dhaka City, Bangladesh.

School dust and soil sampling

The settled dust samples on the windows of the buildings of 11 different schools in Dhaka city were collected using a brush and plastic scoop, while soil samples were collected from the top 3-cm layer after removing the topmost 1-cm layer. A new and clean brush and plastic scoop was used for collecting each sample to avoid cross-contamination. The longitude and latitude of the locations were recorded with GPS, and it should be mentioned here that sampling points cover a major part of the capital city “Dhaka” (Fig. 1). The details of dust and soil sampling locations are represented as

supplementary material in Table S1. The collected all dust and soil samples were air dried naturally in the laboratory for 1 week, then sieved through a 1.0-mm (16 US mesh) nylon mesh (USA Standard Testing Sieve, W.S. Tyler Inc., USA) to remove refuse and small stones. The dried samples were taken in clean zip-lock bags and stored in a desiccator for elemental analysis following the method described elsewhere (Rahman et al. 2021b). The samples were differentiated via labelling using the respective sites in the location. For example, the dust samples were labelled as “D”, whereas soils were assigned “S”.

Analytical procedure

The soil and dust samples were dried at sunlight for 2 days, followed by sieving through a plastic mesh of pore size 0.125 mm (120 meshes ASTM, W.S. Tyler Inc., USA), to remove extraneous matter such as pieces of brick, paving stone and other debris. Afterwards, the samples were dried at about 85 °C in an oven (Vinci Technologies SA - 27 B, France) for 24 h, and sieved through 63 µm pore sized mesh

(230 meshes ASTM, USA). On the other hand, dried soil samples were mechanically grounded using mortar and pestle to get fine grain size and homogeneous soil sample. Afterword, less than 63 µm particle size fraction was collected like dust samples. This size fraction was chosen as those particles are effectively re-suspended (Nicholson 1988; Sehmel 1980), and can be breathed in through the nose or mouth (Ferreira-Baptista and De Miguel 2005). Subsequently, a pellet producer having nearly 10 tons of hydraulic pressure (Specac, UK) was applied for 2–5 min onto 1 g of powder samples (dust and soil) to get 25 mm pellet. The pellet samples were loaded in an auto sample holder for elemental (i.e. Fe, Cu, Zn, As, Pb and Sr) analysis by energy dispersive X-ray fluorescence with a spot size of 40 µm (EDXRF; NEX QC+, Rigaku, Japan) following the technique published in the literature (Ahmed et al. 2007; Hossain et al. 2020; Rahman et al. 2019; Tamim et al. 2016). Limit of detection (LOD) was calculated using three times the square root of the background (Curie 1995). One standard reference material from IAEA 433 (marine sediment) was used for the construction of calibration curves for carrying out elemental analysis in the dust and soil samples. The LOD values and measurement accuracy are presented in Supplementary Table S2.

Pollution assessment

The anthropogenic influence on heavy metal pollution on both the dust and soil samples was assessed using enrichment factor (*EF*), geo-accumulation index (I_{geo}), contamination factor (CF) and pollution load index (*PLI*).

The *EF* was used to assess the degree of metal pollution and to differentiate possible natural or anthropogenic sources. The *EF* was calculated using the following Equation (Barbieri 2016):

$$EF = \frac{(C_x/C_{ref})_{sample}}{(C_x/C_{ref})_{crust}} \quad (1)$$

where $(C_x/C_{ref})_{sample}$ and $(C_x/C_{ref})_{background}$ refer to the ratio of the concentration of a target and reference metal in the analysed samples (soil and dust) and background material, respectively. This study used Fe as a reference metal, and the metal concentrations in the upper continental crust (UCC) as background values (Taylor and McLennan 1985). The *EF* values are generally classified as (i) $EF < 2$ states no to minimal enrichment, (ii) $2 \leq EF < 5$ indicates moderate enrichment, (iii) $5 \leq EF < 20$ indicates significant enrichment, (iv) $20 \leq EF < 40$ indicates a very high enrichment and (v) $EF \geq 40$ indicates an extremely high enrichments (Kowalska et al. 2018).

The I_{geo} , which was initially introduced for contamination assessment of bottom sediment (Muller 1969), is now extensively used to assess the contamination of soil and dust by

comparing the measured metal concentration against background concentration (Kowalska et al. 2018; Othman et al. 2019; Rehman et al. 2020). The I_{geo} was calculated using the following equation:

$$I_{geo} = \log_2 \left[\frac{C_n}{1.5B_n} \right] \quad (2)$$

where C_n is the measured concentration of heavy metals in dust and soil, and B_n is the geochemical background concentration of the same heavy metal adopted from Taylor and McLennan (1985). The constant 1.5 is introduced to minimize the effect of possible variations in the background values, which may be attributed to lithological variations in the sediments. The following classification is given for I_{geo} (Huu et al. 2002; Muller 1969; Othman et al. 2019; Rehman et al. 2020): $I_{geo} < 0$ = practically unpolluted; 0–1 = unpolluted to moderately polluted; 1–2 = moderately polluted; 2–3 = moderately to highly polluted; 3–4 = highly polluted; 4–5 = highly to extremely high polluted and > 5 = extremely high polluted.

The CF is expressed as follows, which gives the degree of increment of a particular heavy metal in soil and dust with regard to the same metal content in the geochemical background:

$$CF = \frac{C_{metal}}{C_{background}} \quad (3)$$

where C_{metal} is the measured concentration of metal in dust and soil, and $C_{background}$ is the background value for the metal. In this study, the metal concentrations in the upper continental crust reported by Taylor and McLennan (1985) were used as background values. The CF values of <1, 1–3, 3–6 and >6 indicate low, moderate, considerable and very high contamination, respectively (Gope et al. 2017).

Finally, each sampling site was evaluated for the cumulative pollution load employing the pollution load index (*PLI*) developed by Tomlinson et al. (1980) as follows:

$$PLI = \sqrt[n]{CF_1 \times CF_2 \times CF_3 \dots \times CF_n} \quad (4)$$

where n is the number of metals studied, and CF is the contamination factor calculated as described in eq. 3 and n is the number of metals. The *PLI* provides simple, but comparative means for assessing a site quality, where a value of $PLI < 1$ denotes perfection; $PLI = 1$ presents that only baseline levels of pollutants are present, and $PLI > 1$ indicates deterioration of site quality (Addo et al. 2012; Kumar et al. 2020).

Health risk assessment model

Non-carcinogenic and carcinogenic human health risks were calculated for both adults and children considering three exposure pathways, namely direct ingestion, inhalation through

mouth and nose and dermal adsorption through exposed skin. For these assessments, health risk models proposed by the United States Environmental Protection Agency were used (US EPA 2002). It should be noted that using USEPA-suggested values in the risk estimation models may add uncertainty in the risk calculation for the Bangladesh school-going children. Integration of local data would give more appropriate risk estimation. However, due to lack of local data, this study adopted the USEPA suggested values.

Non-carcinogenic risk estimation

For non-carcinogenic risk assessment, chronic daily intake (*CDI*), hazard quotient (*HQ*) and hazard index (*HI*) of heavy metals were calculated by using the following equations (US EPA 1989, 1996):

$$CDI_{ingestion} = \frac{C \times IngR \times EF \times ED}{BW \times AT} \times 10^{-6} \quad (5)$$

$$CDI_{inhalation} = \frac{C \times InhR \times EF \times ED}{PEF \times BW \times AT} \quad (6)$$

$$CDI_{dermal} = \frac{C \times SA \times AF \times ABF \times EF \times ED}{BW \times AT} \times 10^{-6} \quad (7)$$

where $CDI_{ingestion}$, $CDI_{inhalation}$ and CDI_{dermal} are the daily exposure amounts through ingestion, inhalation and dermal contact, respectively ($\text{mg kg}^{-1} \text{ day}^{-1}$); C is the heavy metals concentration in soil and dust (mg/kg); $IngR$ is the ingestion rate of soil and dust (mg day^{-1}); $InhR$ is the inhalation rate of dust ($\text{m}^3 \text{ day}^{-1}$); ED is the exposure duration (years); EF is the exposure frequency (days year $^{-1}$); BW is the bodyweight of individual (kg); AT is the average time (days); SA is the surface area of the skin, which comes in contact with dust (cm^2); AF is the skin adherence factor (mg cm^{-2}); PEF is the particle emission factor ($\text{m}^3 \text{ kg}^{-1}$) and ABF is the dermal absorption factor (unitless). All the exposure factors used in health risk calculations are summarized in Supplementary Table S3.

The hazard quotient (*HQ*) and hazard index (*HI*) were used to estimate the non-carcinogenic effects of HMs, and were calculated using the following equations:

$$HQ_{ingestion} = \left(\frac{CDI}{RfD} \right)_{ingestion} \quad (8)$$

$$HQ_{inhalation} = \left(\frac{CDI}{RfD} \right)_{inhalation} \quad (9)$$

$$HQ_{dermal} = \left(\frac{CDI}{RfD} \right)_{dermal} \quad (10)$$

$$HI = \sum_i^n HQ_{ingestion} + \sum_i^n HQ_{inhalation} + \sum_i^n HQ_{dermal} \quad (11)$$

where $HQ_{ingestion}$, $HQ_{inhalation}$ and HQ_{dermal} are the ratio of the chronic daily intake (*CDI*) to the reference dose (*RfD* in $\text{mg kg}^{-1} \text{ day}^{-1}$) of a specific metal for ingestion, inhalation and

dermal adsorption. The hazard index (*HI*) is a cumulative metric that takes into account the combined contribution of *HQs* of metals i to n through three exposure pathways. $HI > 1$ indicates a chance of non-carcinogenic health risk, while $HI \leq 1$ indicates no possible health risk (US EPA 2001).

Carcinogenic risk estimation

The cumulative cancer risk (*CCR*) due to exposure to As from school soil was calculated by adding the cancer risk (*CR*) through ingestion ($CR_{ingestion}$), inhalation ($CR_{inhalation}$) and dermal contact (CR_{dermal}) using Eq. (12). Among the analysed HMs, As was evaluated for cancer risk as the cancer slope factor (*CSF*) values for three exposure routes are available only for As:

$$\begin{aligned} CCR = & CR_{ingestion} + CR_{inhalation} + CR_{dermal} \\ = & (LADD_{ingestion} \times CSF_{ingestion}) \\ & + (LADD_{inhalation} \times CSF_{inhalation}) \\ & + (LADD_{dermal} \times CSF_{dermal}) \end{aligned} \quad (12)$$

where $CSF_{ingestion}$, $CSF_{inhalation}$ and CSF_{dermal} are the cancer slope factors (kg day/mg) of As through ingestion, inhalation and dermal adsorption, respectively. The *LADD* is the lifetime average daily dose (mg/kg/day) and defined by the weighted average for each exposure routes as given below (Ferreira-Baptista and De Miguel 2005; US EPA 1996, 2002):

$$\begin{aligned} LADD_{ingestion} = & C \times \frac{EF}{AT} \\ & \times \left(\frac{IngR_{child} \times ED_{child}}{BW_{child}} + \frac{IngR_{adult} \times ED_{adult}}{BW_{adult}} \right) \times 10^{-6} \end{aligned} \quad (13)$$

$$\begin{aligned} LADD_{inhalation} = & C \times \frac{EF}{AT \times PEF} \\ & \times \left(\frac{InhR_{child} \times ED_{child}}{BW_{child}} + \frac{InhR_{adult} \times ED_{adult}}{BW_{adult}} \right) \end{aligned} \quad (14)$$

$$\begin{aligned} LADD_{dermal} = & C \times \frac{EF \times ABF}{AT} \\ & \times \left(\frac{SA_{child} \times ED_{child}}{BW_{child}} + \frac{SA_{adult} \times ED_{adult}}{BW_{adult}} \right) \times 10^{-6} \end{aligned} \quad (15)$$

There are no absolute criteria for the acceptable number of additional cancer over a lifetime period. However, the USEPA generally adopt one additional case of cancer in 1 million (i.e. 1.0E-6) as a management goal for the government to suggest the point at which management decisions should be taken. The cancer risk surpassing 1.0E-4 (1 case of cancer in 10,000) is considered as unacceptable (US EPA 2005). The *RfD* and carcinogenic slope factors (*CSF*) values are provided in Supplementary Table S4.

Statistical analysis

Multivariate statistics such as Pearson correlation, K mean cluster analysis using Euclidean distance and hierarchical cluster analysis by applying Euclidean distance and Ward's cluster methods were performed to elucidate the possible sources of metal(oid)s in school dust and soil. All statistical analyses were performed using the statistical package R (version 4.0.4).

Results and discussion

Metal concentrations in dust and soil samples

The measured concentration of metal(oid)s in road dust and soil samples in different school compounds in Dhaka city are presented in Supplementary Table S5. Only the metals of environmental concerns (Cu, Pb, Zn and As) are graphically shown in Fig. 2. The mean concentration of heavy metals in both school dust and soil showed the following decreasing order: Ca > Fe > K > Ti > Sr > Zn > Zr > Rb > Cu > Pb. The concentration of Fe, Cu, Zn, Pb, Sr, K and Ca were higher in the dust as they are a complex mixture originating from both natural and anthropogenic sources (Bi et al. 2013; Saeedi et al. 2012). The concentration of Ti and Rb were higher in the soil in most of the school premises (Supplementary Table S5).

According to paired student *t* test, the metal concentrations in both dust and soil varied significantly ($p < 0.05$) among different schools. The concentration of Zn was as high as 330 mg/kg in school dust samples at Azimpur Govt. Girls School & College (ASC), while the highest concentration of Cu was found to be 91.1 mg/kg in school dust at Rajuk Uttara Model College (RUC) (Supplementary Table S5). The highest Fe, Pb and Ti in school dust was found at 2.7%, 136 mg/kg, and 2896 mg/kg, respectively at Dhaka Govt. Muslim High school (DMS). Besides, the highest concentration of dust Rb and Ca were 98.2 mg/kg and 13.1%, respectively, at Dhanmondi Govt. Boys School (DBS). The dust from the National Bangla High school (NBS) contained the highest level of Sr, Zr and K. In the same way, the lowest concentration of metals in school dust was sparsely distributed among the schools but mostly concentrated at Motijheel Model School & College (MSC) (Supplementary Table S5).

For the soil samples, the highest concentration of most of the investigated metals was concentrated at Azimpur Govt. Girls School & College (ASC) (Supplementary Table S5). For example, the highest concentration of Zn, As, Pb, Sr, K and Ca was found in soil samples at this school. At Residential Model College (RMC), the highest concentration of soil Fe, Cu and Rb were 30,590, 59.8 and 149 mg/kg, respectively. Besides, the lowest concentration of metals in school soil

mostly prevailed at Mirpur Govt. Primary School (MGS). Interestingly, the Pb content in the soil as well dust was at the lowest value in this school premises. Overall, soil from Azimpur Govt. Girls School & College (ASC) and Residential Model College (RMC) were highly contaminated with most of the metals, while dust contamination by metals varied from school to school (Supplementary Table S5).

A comparison of our metal(oid) concentrations with other similar studies worldwide is presented in Table 1. In dust, the mean concentrations of Cu (58.5 mg/kg), Zn (187 mg/kg) and Pb (57.0 mg/kg) were ~ 2 times higher than their respective UCC values. The mean Cu concentration for this study is almost similar with the previously reported result in road dust (49.8 mg/kg) in Dhaka city, Bangladesh (Rahman et al. 2019), but much lower than street dust from Kolkata, India (92.0 mg/kg) (Kolakkandi et al. 2020). Conversely, the average Zn concentration in school dust is lower than road dust from Dhaka city, Bangladesh (239 mg/kg) (Rahman et al. 2019a) and Kolkata, India (289 mg/kg) (Kolakkandi et al. 2020). A good number of Zn-releasing industries/mills are in operation in Dhaka City, which may act as a continuous source of Zn in the environment, for instance, galvanized steel road equipment such as crash barriers, road signs and lamp posts. As they corroded and tire wear, Zn compound in rubber may continuously release into the environment (Rahman et al. 2019; Yuen et al. 2012). In addition, small metal smelters and steel re-rolling mills in the old Dhaka City may also be the sources of high concentration of Zn in the soil and dust samples (Rahman et al. 2021a). Road dust Pb in Dhaka was reported to be 18.9 mg/kg (Rahman et al. 2019), while its average concentration in school dust (57.0 mg/kg) in our study was about three times higher. According to this study, a significant amount of Pb is still present in the school dust and soil sample of Dhaka city, which might have different sources (i.e. paint, battery) other than vehicle emissions (Rahman et al. 2019). It should be mentioned here that about 56% of paints purchased from China contain Pb equal to or larger than the Chinese regulation limits (600 ppm) (Yuen et al. 2012). In addition, indoor and outdoor school dust in Malaysia indicated lower Pb content (indoor dust 10.6 mg/kg and outdoor dust 18.2 mg/kg) than school dust in this study. However, Pb content in our study agreed well with school and park dust in Jiaozuo (Henan Province, China) (Table 1). Lithogenic Ca concentration in street dust from Kolkata, India, was 5.68% (Kolakkandi et al. 2020), which is similar to this study (5.95% in dust and 5.49% in soil). These elevated levels of Ca in school dust and soil could be related to cement and lime-based mortar dust from building and transportation construction. In contrast to dust, the concentration of Cu, Zn, Rb and Zr in soil was 39.2, 75.8, 116 and 247 mg/kg, respectively, which are a bit higher than their corresponding UCC values (Table 1). Besides, the measured mean As concentration in soil samples (16.3 mg/kg) was about 11 times higher than the

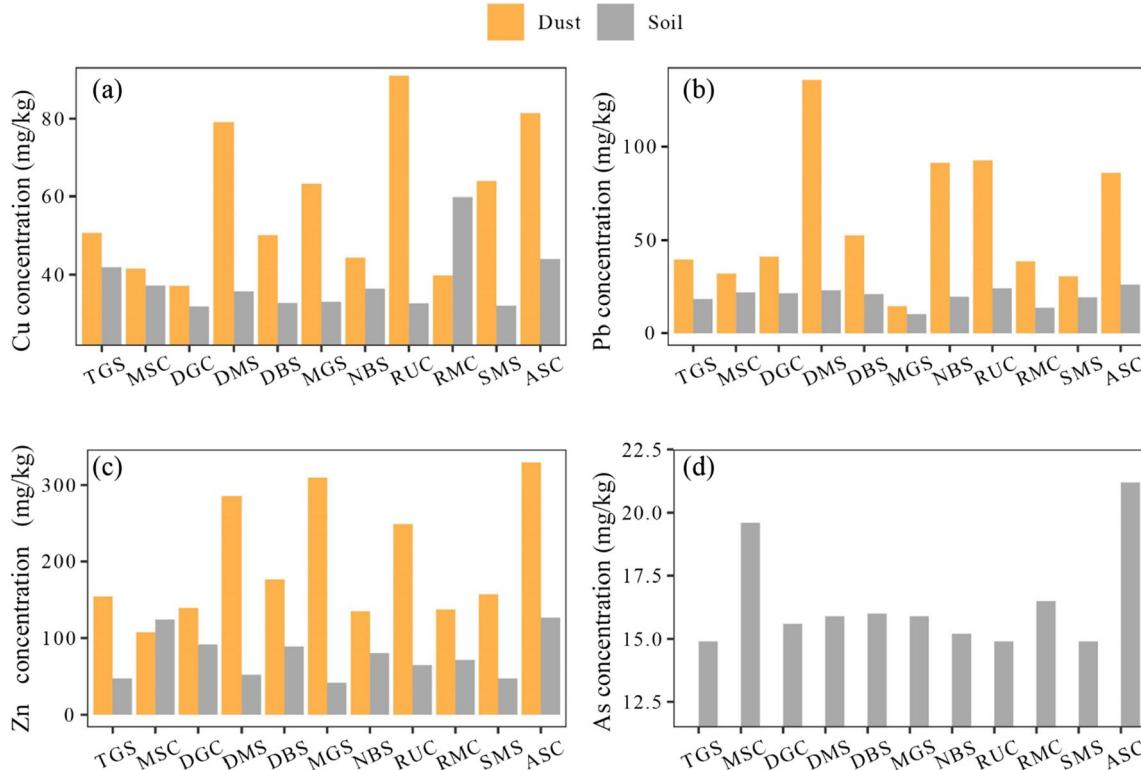


Fig. 2 Bar plots showing the variation of **a** Cu, **b** Pb, **c** Zn and **d** As in dust and soil of different schools in Dhaka city, Bangladesh. Refer to Supplementary Table S1 for school acronyms used in the y axis. Note

that the concentration of only the elements of environmental concern are graphically shown in this figure and the concentration of rest of the measured elements are tabulated in Supplementary Table S4

corresponding UCC value (1.5 mg/kg). It was also higher in school soil in Kuala Lumpur, Malaysia, and playground soil in Beijing, China (Table 1). However, school and park soil of Jiaozuo, China, contained ~4 times higher As concentration than this study (Han et al. 2020). The mean concentration of soil Pb found in this study (19.7 mg/kg) was in line with the UCC value (20 mg/kg), while ~2 times lower than school soil in Malaysia and playground soil in Beijing, China (Table 1). It has been suggested that Cu, As and Zn, might be mainly associated with industrial activities and heavy traffic density considering the sampling sites.

Geochemical indices for pollution assessment

The level of environmental pollution due to the presence of metals in road dust and soil in different school compounds in Dhaka city was evaluated using different pollution assessment indices including EF, I_{geo} , CF and PLI.

Enrichment factor (EF)

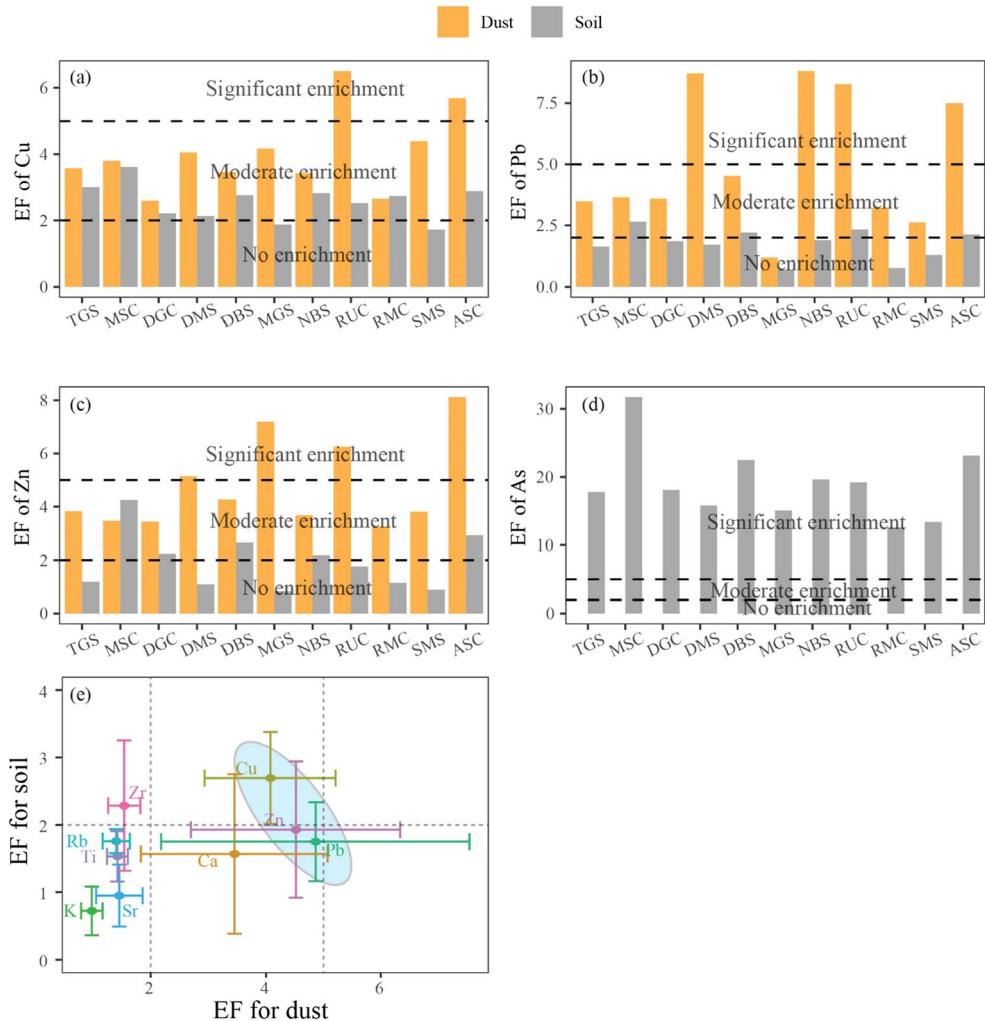
The EF values for Cu, Pb and Zn showed significant variability across the sites, and between soil and dust samples (Fig. 3). The dust samples were more enriched relative to soil samples and showed moderate to significant enrichment in almost all the sites (Fig. 3e). Dust is more enriched with metals than soil

samples in the urban environment as they are a complex mixture originating from natural sources such as atmospheric deposition, soil erosion and resuspension of deposited particles (Amato et al. 2010) as well as anthropogenic activities such as vehicular traffic emission, power generation, fossil fuel combustion, industrial exhausts and construction processes (Bi et al. 2013; Saeedi et al. 2012). In the street dust of Kolkata (a megacity in India), the EFs of Cu, Zn and Pb were in the following ranges 3.8–5.8, 3.4–4.9 and 3.8–12.0, respectively (Kolakkandi et al. 2020), which is consistent with this study (Supplementary Table S6). The enrichment of soil and dust with Cu in a busy traffic city like Dhaka is likely to be associated with leakage of fuel and oil from vehicles, and the wearing of engine parts, brake lining pads, fibres and other vehicle accessories with Cu-based metal alloys (Gope et al. 2018). Although the uses of leaded gasoline in Bangladesh have been phased out in 1999 (Biswas et al. 2003), Pb from previously combusted fuel may have been mixed with the soil and dust, and have not removed completely from the environment yet. Besides, the uses of leaded paint in schools may have also enriched the soil and dust with Pb in school premises as was reported for old built-up areas (Hunt 2016; Laidlaw et al. 2017). The enrichment of Pb in street dust of Dhaka city has recently been reported by Rahman et al. (2019). Excess Zn may have derived from the mechanical corrosion and wearing of car parts because this element is used

Table 1 Concentrations of metal(oids) (mean \pm standard deviation, mg/kg) in the school dust and soil in Dhaka and comparison with those reported from other locations in the world and UCC values

Sample type, study area (number of samples)	K	Ca	Rb	Sr	Fe	Cu	Pb	Zn	As	Ti	Zr	References
School dust, Dhaka City (11) 2063	59,509 \pm 28,462	88.5 \pm 7.5 64.2	287 \pm 2790	20,112 \pm 36.1	58.5 \pm 17.8 49.7 \pm 12.6	57.0 \pm 19.0 \pm	187 \pm 84.7 239 \pm 144	— 8.09 \pm 1.75	2431 \pm 338	168 \pm —	— —	This study (Rahman et al. 2019)
Street dust, Dhaka City (22)	—	—	—	—	141 \pm 27.7	—	14 \pm 6.59 40,334 \pm 4061	25 \pm 4.81 92 \pm 47	65 \pm 15.3 128 \pm 127	4 \pm 0.34 289 \pm 95	— —	286 \pm 105 (Ahmed et al. 2007)
Street dust, Kolkata city, India (57)	—	56,810 \pm 16,092	—	—	—	—	40 \pm 22.7 40,334 \pm 4061	259 \pm 264 233 \pm 207	233 \pm 207 27.8 \pm 1.72	— —	— —	(Kolakkandi et al. 2020)
School dust, Shiraz megalcity, Iran (32)	—	—	—	—	—	—	40 \pm 22.7 195	116	230	1053	7.51	— —
School dust, Lahore, Pakistan (40)	—	—	—	—	—	—	—	—	—	—	—	(Rehman et al. 2020)
Indoor school dust, Kuala Lumpur, Malaysia (NM)	2.66 \pm 1.13	38.5 \pm 35.0	—	—	219 \pm 45.7	10.3 \pm 6.97	10.6 \pm 141 \pm 8.3	289 \pm 95	5.23 \pm 1.05	— —	— —	(Othman et al. 2019)
Outdoor school dust, Kuala Lumpur, Malaysia (NM)	2.58 \pm 0.64	25.1 \pm 9.66	—	—	653 \pm 29.2	6.94 \pm 2.98	4.57 18.2 \pm	386 \pm 10.1	10.5 \pm 4.91	— —	— —	(Othman et al. 2019)
School and park: dust, Jiaozuo, Henan Province, China (41)	—	—	—	—	—	49.9 \pm 20.92	55.3 \pm 48.5	374 \pm 325	23.08 \pm 4.50	— —	— —	(Han et al. 2020)
Children's playground dust, Beijing, China (71)	—	—	—	—	30,740 \pm 9,771	52.1 \pm 38.7	80.3 \pm 56.0	— 39.2 \pm 9.14	— 19.7 \pm 75.7 \pm 16.3 \pm 2.01	10.02 \pm 2708 \pm 649	— — —	(Jin et al. 2019)
School soil, Dhaka City (11) (NM)	11,875 \pm 5833	25,626 \pm 16,853	116 \pm 17.3	186 \pm 71.7	4534	4.40 28.5	— —	— 17.9 \pm 0.92	39.6 \pm 2.91	— 99.5 \pm 65.0	247 \pm 82.3	This study (Othman et al. 2019)
School soil, Kuala Lumpur, Malaysia (NM)	236 \pm 54.1	233 \pm 65.3	—	—	2695 \pm 135	17.9 \pm 0.92	39.6 \pm 2.91	66.5 \pm 9.2	10.8 \pm 0.72	— —	— —	(Othman et al. 2019)
School and park: soil, Jiaozuo, Henan Province, China (41)	—	—	—	—	—	—	20.9 \pm 5.19	30.9 \pm 18.3	57.7 \pm 46.9	— —	— —	(Han et al. 2020)
Children's playground soil, Beijing, China (71)	—	—	—	—	27,738 \pm 4,604	43.4 \pm 17.6	36.6 \pm 40.4	— —	15.5 \pm 2.24	— —	— —	(Jin et al. 2019)
Upper continental crust (UCC) value	28,000	30,000	112	350	35,000	25	20	71	1.5	3000	190	(Taylor and McLennan 1985)

Fig. 3 Enrichment factors (EF) of **a Cu**, **b Pb**, **c Zn**, and **d As**. The cross-plot (**e**) showing the enrichment factors for soil and dust in different schools of Dhaka city. Error bars in **e** represents standard deviation



in the production of brake lining, lubricating oil clumps and cylinder head gasket (Trujillo-González et al. 2016). The oxidation of lubricating oil, that contains Zn compounds as antioxidant and dispersant, due to exposure to air can release various organic acids that can subsequently corrode the metallic part of the vehicle and release Zn to the environment (Gope et al. 2018).

A striking feature of this study was the very high enrichment of soil with As in the school areas (Fig. 3d). The EF values of soil As ranged from 13 to 32 (highest values recorded at Motijheel Model School and College), indicating the significant anthropogenic influence of As. It is well known that Bangladesh is one of the hotspots of geogenic As pollution in the world, particularly in groundwater (Bhattacharya et al. 2009; Nickson et al. 2000; Saha et al. 2020; Saha and Rahman 2020). Arsenic contaminated groundwater has the potential to increase soil As over time (Dittmar et al. 2010; Panaullah et al. 2009). In the urban areas, building constructions require excavation of a large amount of subsurface sediments that could be a potential source of As release in urban soil as a result of alteration in the environmental condition of

excavated land (Molinari et al. 2013; Tabelin et al. 2012a, b). Like other cities in the world, the densely populated Dhaka city also experiences a large number of construction activities. As a result, natural sources of As and human activities pose a synergistic effect on substantial As release in Dhaka city, Bangladesh (Fig. 3). However, the EF values for K, Rb, Sr, Ti and the majority of Zr in the soil, as well as dust, were below 2, indicating the natural origin of these metals (Supplementary Table S6).

Geo-accumulation index (I_{geo})

The geo-accumulation index (I_{geo}) of the investigated metals in school dust and soil was calculated following Eq. (2), and detailed results for each metal could be found in Supplementary Table S7. The I_{geo} values of Cu, Pb and Zn in school dust were mostly in the categories of ‘unpolluted to moderately polluted (i.e. $I_{geo} = 0 - 1$)’ with ‘moderately polluted (i.e. $I_{geo} = 1 - 2$)’ dusts around Azimpur Govt. Girls School & College, Rajuk Uttara Model College and Dhaka Govt. Muslim High School. Although the school dust samples

were polluted with Cu, Pb and Zn, school soils were ‘practically unpolluted (i.e. $I_{geo} < 0$)’ with respect to these metals. Based on I_{geo} of As, the school soils ($I_{geo} = 2.7$ to 3.2) were ranged between ‘moderately to highly polluted ($I_{geo} = 2 - 3$)’ and ‘highly polluted ($I_{geo} = 3 - 4$)’ categories. Such high contamination of soil As was also suggested by *EF* analysis. The I_{geo} values of the rest of the metals in soil and dust were < 0 , which indicated practically uncontaminated condition (Supplementary Table S7).

Contamination factor (CF) and pollution load index (PLI)

The values of CF and *PLI* for different metals in dust and soil samples were calculated following Eqs. (3) and (4), and the results are presented in Supplementary Table S8. The CF values of metals, particularly Cu, Pb and Zn varied significantly between school dust and soil samples, and across the school locations. In general, dust samples were considerably contaminated with Cu, Zn and Pb, while the soils were highly contaminated with As. The CF values of As in school soil were > 6 (ranged from 9.9 to 14.1 with a mean value of 10.9) and fall in this category of ‘very high contamination’ (Mmolawa et al. 2011). As it was mentioned earlier, this high level of As in school soil was likely to be originated mostly from natural sources and anthropogenic influence has triggered the release of As into urban soil. At some schools, the CF values of Cu, Zn and Pb in dust samples were higher than 3, indicating ‘considerable contamination’ (Supplementary Table S8). For example, the CF values of Cu, Zn and Pb in school dust were more than 3 at Azimpur Govt. Girls School & College (ASC), Rajuk Uttara Model College (RUC) and Dhaka Govt. Muslim High school (DMS). It is noteworthy that these three locations are highly populated and dense traffic areas that may have contributed to this elevated level of Cu, Zn and Pb contamination in school dust and soil. However, the rest of the metals were in the category of ‘low contamination’ with CF mostly < 1 (Supplementary Table S8). The assessment of contamination based on CF was consistent with the results of *EF* and I_{geo} of the corresponding metal(oid)s.

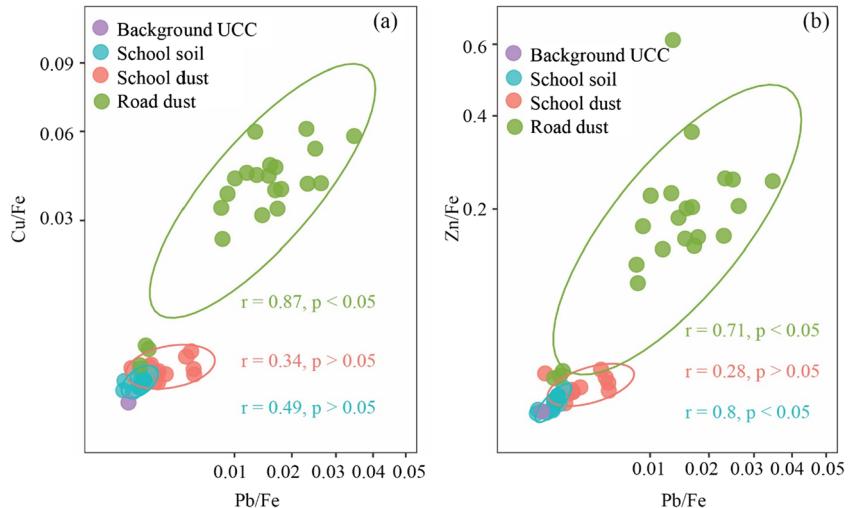
To further assess the quality of school dust and soil, *PLI* was calculated that give the pollution level at each sampling point (see Supplementary Table S8). The school dust at every sampling point was more polluted than school soil. The *PLI* value greater than unity indicates deterioration of site quality by metal(oid)s. In dust samples, the *PLI* > 1 was observed in three contaminated sites, including Azimpur Govt. Girls School & College (ASC), Rajuk Uttara Model College (RMC) and Dhaka Govt. Muslim High school (DMS). However, in soil samples, the highest *PLI* value (~2.8) was found at Azimpur Govt. Girls School & College. Overall, the dust and soil samples at Azimpur Govt. Girls School & College were the highest polluted site among the studied schools in Dhaka city, and such high pollution was possibly

related to the dense population and increased traffic emission in the Azimpur area.

The probable pollution sources of metals in soil and dust

The plots of Fe normalized Cu vs. Pb and Zn vs. Pb from school soil and dust (this study), and road dust (Rahman et al. 2019), as well as the upper continental crust (Taylor and McLennan 1985) as a background, show their distribution positions and a gradient of the magnitude of pollution (Figs. 4a, b). The ratios of Cu/Fe and Zn/Fe against Pb/Fe in school soils plotted closer to the background value and progressively distributed further apart, while the school dust samples were characterized as having higher Cu/Fe, Zn/Fe and Pb/Fe than that of background value with partial mixing with school soils. The roadsides dust in Dhaka formed a distinctive cluster with higher ratios of Cu/Fe, Zn/Fe and Pb/Fe than the school dust and soil in Dhaka. The higher metal ratios in road dust followed by school dust relative to the values in the background as well as school soils indicate a greater influence of vehicular traffic sources in dust samples (Fig. 4). Vehicular traffic is considered as a significant contributor to metals in dust due to the rapid increase in the number of vehicles in Dhaka city with enormous traffic congestion (Rahman et al. 2019). For example, abrasion of the tire, uses of lubricants and corrosion of vehicular parts may lead to Cu and Zn pollution, while the existence of previously combusted leaded gasoline may play the significant role in polluting the dust with Pb. However, the closeness of metal ratios in the soil to the background values indicate the coexistence of geogenic and anthropogenic sources (e.g., vehicular emissions and abrasion of anti-corrosive paints used on iron fence and other infrastructures in school areas) of Cu, Zn and Pb (Dong et al. 2017; Lu et al. 2009). Besides, the K-means cluster analysis using all the analysed metals (except As due to lack of data in school dust) shows an apparent disjunction between soil and dust samples from the schools, suggesting the differential degree of anthropogenic influence on the abundance of metals hosted in soil and dust (Fig. 5a). The soil samples from the Azimpur Govt. Girls School & College (ASC), Dhanmondi Govt. Boys School (DBC) and Motijheel Model School & College (MSC) fall in the boundary of dust clusters, suggesting relatively higher contamination of those soils relative to other schools. Furthermore, hierarchical cluster analysis differentiated the analysed metals among three groups wherein Cu, Pb and Zn were clustered in the same group, suggesting their identical source of origin, possibly from traffic-related sources as discussed above (Fig. 5b). The second group encompassed Zr, Fe, Ti and Rb, while the third group consisted of K, Sr and

Fig. 4 Bivariate plots showing the distribution of Fe normalized Cu and Pb (a) and Zn and Pb (b) in school soil and school dust (this study) and road dust (Rahman et al. 2019) in Dhaka city, and the upper continental crust (Taylor and McLennan 1985)



Ca. Based on the enrichment calculation (discussed in the “Enrichment factor (EF)” section), we hypothesised that the metals of the second cluster have been originated from the natural sources with minimal to no contribution from anthropogenic sources in the school areas, while the metals in the third cluster have been influenced by industrial activities such as cement and lime-based mortar dust from the widespread buildings, roads, flyovers and transportation construction in Dhaka city (Appleton and Cave 2018; Itkin et al. 2016).

Evaluation of human health risk

Non-carcinogenic health risks

The non-carcinogenic health risks of adults and children due to exposure to Fe, Cu, Zn, Pb and Sr in dust and soil samples from three pathways (ingestion, inhalation and dermal adsorption) were estimated in terms of hazard quotient (HQ) and hazard index (HI) following Eqs. (5) to (11). The results for adult and children are presented in Supplementary Tables S9

and S10, respectively. The graphical representation of HQ values for both adult and children exposed to metals in dust and soil samples can be found in Supplementary Figs. S1 and S2. This study revealed that the HQ values of different metal(oid)s in dust and soil samples varied significantly, and the values were consistently higher for children than the adults regardless of metal(oid)s and exposure pathways (Supplementary Figs. S1 and S2). Ingestion was the dominant route of exposure to metal(oid)s followed by dermal contact and inhalation. Similar findings were reported in street dust from Dhaka city, Bangladesh (Rahman et al. 2019), and surface dust from Tangshan city, China (Cui et al. 2020). Among the analysed metal(oid)s in the dust samples, Pb exhibited the highest HQ value (3.3E-01 for children and 3.5E-02 for adults) through ingestion followed by Cu, Zn and Fe for both adults and children (Supplementary Fig. S1). For soil, the highest HQ values were observed for Pb (6.2E-02 for children) through ingestion and the lowest for Zn (1.8E-8 and 3.3E-8 for adults and children, respectively) through soil inhalation (Supplementary Fig. S2). Overall, all the investigated metal(oid)s in school dust and soil showed HQ values lower

Fig. 5 a K-means cluster analysis showing the formation of two separate clusters for soil and dust samples. b Hierarchical cluster analysis showing the grouping of 10 analysed metal(oid)s into three clusters. Note that the sample labels start with prefix S and D represent soil and dust, respectively

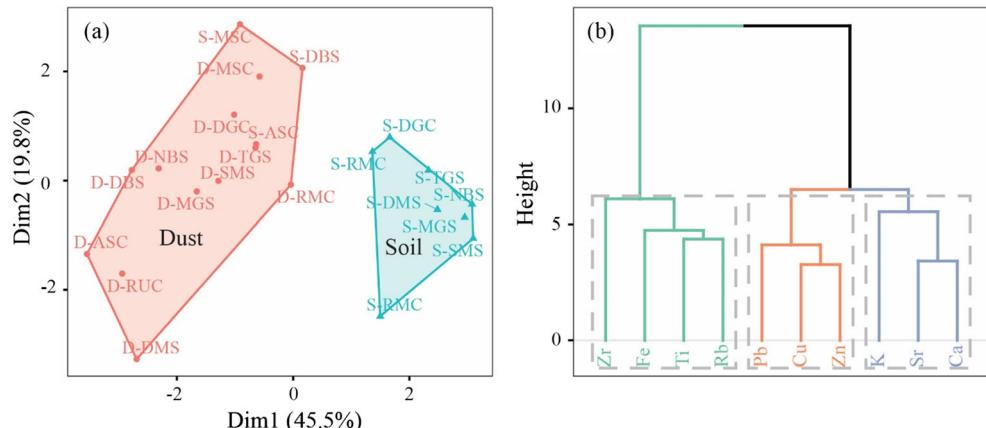
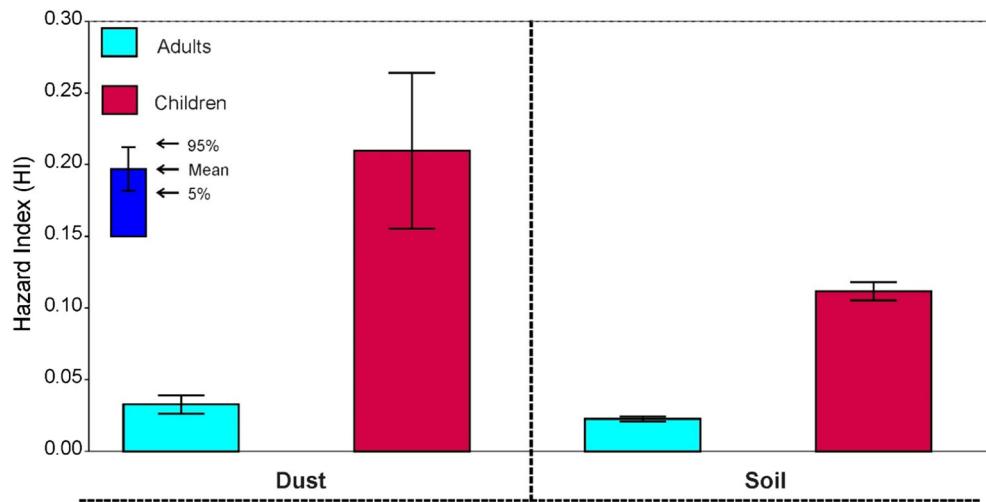


Fig. 6 Estimated hazard index (*HI*) through three exposure pathways of school dust and soil in Dhaka city, Bangladesh



than the threshold value of 1 (Supplementary Tables S9 and S10), suggesting no possible non-carcinogenic human health risk.

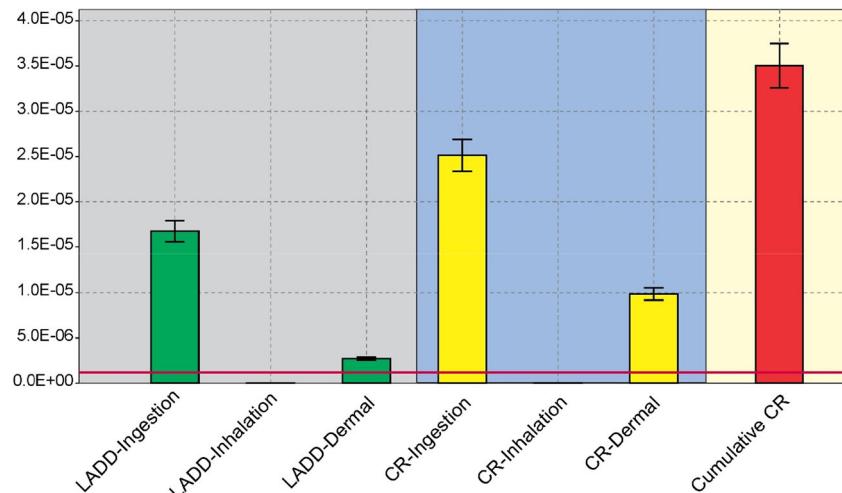
The estimated hazard index (*HI*) values for dust were higher than soil for both adults and children (Fig. 6). The highest *HI* value of 0.43 was observed in the dust for the children, and the lowest was 0.02 in the soil for adults in the study area. Exposure to Pb and Fe were identified as two major contributors to the cumulative hazard index, i.e. *HI* calculated for dust and soil. The *HI* values for all cases remained below unity, indicating a low or no non-carcinogenic human health risk in the study area (Supplementary Tables S9 and S10). However, children were more susceptible to non-cancer risk than adults. It was reported that elevated Pb concentration in children might result in lower Intelligence Quotient (*IQ*), learning disorders and behavioural problems (Nevin 2000; Nigg et al. 2008). Furthermore, Pb is responsible for several diseases such as motor neuron disease (Laidlaw et al. 2015; Santurtún et al. 2016), retardation in children development (Earl et al. 2016),

mental illness (Guilarte et al. 2012) and brain cancer (Wu et al. 2012). The health risks of school-going children associated with elevated Pb concentration are of great concern in Dhaka city, Bangladesh.

Carcinogenic health risk from As in school soil

Considering lifetime exposure, lifetime average daily dose (*LADD*) and carcinogenic risk (*CR*) through ingestion, inhalation and dermal adsorption were calculated for As in school soil since among the investigated metal(loid)s, only As has the CSF value and As concentration was measured only in soil samples in this study (Supplementary Table S11). Figure 7 depicts the *LADD* and *CR* for individual exposure pathways and cumulative *CR* (*CCR*) for the sum of *CR* from all exposure routes. The highest *LADD* was observed through soil ingestion, followed by dermal contact and inhalation. Similarly, the *CR* was the highest for soil ingestion (3.3E-5) followed by dermal adsorption (1.3E-5) and inhalation (2.1E-8). The *CR* of As through soil ingestion (average 2.5E-05;

Fig. 7 The Lifetime Average Daily Dose (*LADD*), Carcinogenic Risk (*CR*) and Cumulative Carcinogenic Risk (*CCR*) for As in school soil from Dhaka city in Bangladesh (the purple line indicates lower limit of acceptable value for cancer risk)



range 2.3E-05 to 3.3E-05) and dermal adsorption (average 9.9E-06; range 9.0E-06 to 1.3E-05) exceeded the value 1.0E-6, which is considered as a precautionary criterion internationally at which the government should take the risk management decision (Rahman et al. 2019). The CCR value of As ranged from 3.2E-05 to 4.6E-05 with a mean value of 3.5E-5 (Fig. 7), suggesting significant exceedance of the precautionary limit of 1.0E-6. It has been reported that long-term exposure of As can cause multi-system carcinogenicity such as skin, lung, urogenital tract, gastrointestinal and other tissues (Lansdown 2013). Therefore, it has been suggested that higher As concentration in school soil can be considered as a cancer risk in Dhaka city. Other studies on street dust from Dhaka city also indicated a potential cancer risk for As (Rahman et al. 2019).

Implications for the management of health risk and policymaking

For the first time, this study conducted a detailed assessment of metal(oid)s pollution in school environments and estimated the possible health risks to the population, especially for the school-going children in Dhaka city. It is evident that health risks are mostly related to traffic- and construction-originated metals in the school dust and soil. Therefore, vehicular and construction-related emission should be reduced, and the school campuses should be relocated to greener areas, far from high traffic areas. Besides, construction materials, including furniture and paints, should be free from heavy metals and have lower emission profiles. If the schools cannot be relocated, air ventilation system, low-cost air quality monitoring devices and effective air filtration devices should be installed to reduce the exposure of children to soil and dust particles (Oliveira et al. 2019). Tree plantation around the schools may reduce soil and dust-related heavy metal(oid)s pollution as the trees are capable of capturing particulate matters and accumulate metals in an environmentally friendly manner (Serbulu et al. 2012; Zhang et al. 2017). Besides, spraying water in the school premises, especially in the dry winter season, may reduce the dispersion of fine dust particles.

Conclusions

The present study analysed the concentration of metal(oid)s in dust and soil samples collected from 11 school premises in the megacity Dhaka in Bangladesh to identify the possible sources of pollution in the school areas. According to the pollution evaluation indices, dust samples in almost all the sites showed moderate to significant enrichment in terms of Pb, Cu and Zn. School soil was strongly contaminated with As, along with unpolluted to moderately polluted with Cu at some schools. Multivariate analysis indicated that

anthropogenic influences like high motor vehicle traffic and industrial activities are responsible for the enrichment of Pb, Cu and Zn. The non-cancer health risk estimated based on *HI* suggested low or no non-carcinogenic human health risk. Ingestion of Pb from school dust and soil contributed the highest to the estimated non-carcinogenic health risk. However, CCR value of soil As greatly exceeded the precautionary limit of 1.0E-6, suggesting that the Bangladesh government should take actions for the management of As-related cancer risk through exposure to the soil in school premises in Dhaka city, Bangladesh. The vehicular emission and construction activities in Dhaka city should be reduced to safeguard the health of school-going children.

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Authors' contributions SR designed the experiments, collected samples, performed analysis and reviewed drafts. SK was a major contributor in writing the manuscript. MN helped in drafting the manuscript. NS analysed data, designed the manuscript framework and significantly contributed to writing, reviewing and editing the manuscript.

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Availability of data The datasets generated and/or analysed during the current study are available in the Supplementary Information.

Declarations

Ethics approval and consent to participate Not applicable.

Consent to publish Not applicable.

Competing interests The authors declare that they have no competing interests

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