



# Potentially toxic elements in street dust from an urban city of a developing country: ecological and probabilistic health risks assessment

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## Abstract

Anthropogenic activities in and around the urban highways followed by aerodynamic processing generate street dusts, which can cause adverse health effects through different exposure pathways. Hence, considering the high degree of industrialization, concomitant unplanned urbanization, and rapid demographic augmentation, street dust samples from an urban city (Gazipur, Bangladesh) were investigated in terms of potentially toxic elements (using ICP-MS) to evaluate their ecological and health risks. Mean concentrations ( $\pm$  SD) of lead (Pb), copper (Cu), chromium (Cr), cadmium (Cd), zinc (Zn), nickel (Ni), and arsenic (As) in the analyzed air-dried samples were  $40.9 \pm 13.6$ ,  $44.9 \pm 15.4$ ,  $83.3 \pm 19.0$ ,  $9.1 \pm 5.4$ ,  $239.1 \pm 34.7$ ,  $33.5 \pm 10.4$ , and  $2.1 \pm 0.8$  mg/kg, respectively with heterogeneous distribution which were 0.2 (As) to 82.7 (Cd) times higher than the available internationally recommended limits. Element-specific environmental indices revealed that contamination levels followed the descending order as Cd > Zn > Cu > Pb > Cr > Ni > As, whereas individual ecological risks followed the descending order as Cd > Cu > Pb > Ni > Zn > Cr > As. Sampling site-specific composite indices indicated that sampling sites with high loadings of traffic, population, industrialization, and urbanization were mostly polluted. Multivariate statistical approaches also deduced the similar origins of the studied elements. In terms of the investigated elements, the study site possessed high potential ecological risks, although non-carcinogenic and carcinogenic risks through different pathway's exposures seem insignificant, where children are more vulnerable than adults.

**Keywords** Potentially toxic elements · Street dusts · Gazipur city, Bangladesh · Spatial distribution · Ecological and health risks

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## Introduction

Street dust is formed by natural geochemical processes (e.g., soil formation, weathering of minerals, rocks, etc.) and variety of anthropogenic causes (e.g., building and dismantling doings, transportation, deterioration of zinc-plated metal structures/safety fences, manufacturing facilities, burning of fossil-fuel, etc.) and collects in the metropolitan surroundings from atmospheric depositions along impervious streets (Trojanowska and Świetlik 2019; Idris et al. 2018). Moreover, car traffic has a disproportionately large effect on the urban environment's possible toxic factor emissions (Charlesworth et al. 2003). Therefore, one of the most powerful and profound measure of urban terrestrial emissions caused by possible toxic elements is street dust, which is both a sink and a source of toxic pollutants (Urrutia-Goyes et al. 2018; Huang et al. 2016). However, urban street dust is one of the consistent indicators for the quality of air in particular (Idris et al. 2018). Unplanned urbanization and haphazard industrialization along with rapid population growth and environmental pollution by potential toxic elements are becoming more alarming in developing countries like Bangladesh (Kabir et al. 2020a, 2020b, 2020c). Due to the expansion of urban areas, impervious surfaces such as roads or streets are increasing; in contrast, open spaces are decreasing significantly. Automobiles typically emit a plethora of toxins, as well as potentially harmful elements, from the combustion of fuel and consumable materials for instance chemicals, lubricants, rubber, and brake pads (Prokofeva et al. 2017). Consequently, researchers have given particulate deposits a lot of thought for environmental control of potentially harmful elements (Ustaoğlu and Islam 2020; Ustaoğlu and Tepe 2019; Pan et al. 2017; Mathur et al. 2016; Ma et al. 2016).

Metropolitan street dust collects different contributions of heavy metals from variety of versatile and fixed sources, for instance, vehicular discharge, power generation plants, mechanical plants, and waste cremation (Rahman et al. 2019; Ahmed and Ishiga 2006; Duan et al. 2017; Kamani et al. 2017; Delibašić et al. 2020; Suryawanshi et al. 2016). Various modern metallurgical cycles are the preeminent admission sources of Cd, Ni, Zn, Cu, and As in metropolitan earthly environment (Men et al. 2018). Automobiles are the most important contributors of Pb, Cr, Cu, Zn, Ni, Cd, and Fe to urban street dusts (Delibašić et al. 2020). The burning of petroleum derivatives is the most significant anthropogenic source of Mn, Mo, Co, Ni, Be, Cr, Cu, Se, V, As, Sn, Sb, and Hg (Al-Momani 2007). Copper and zinc were also contained in high concentrations in street dust from both mechanical and vehicular traffic areas (Shi et al. 2008). Lubricating oils, brake cushions, tires, wearing of hardware parts, fuel burning, vehicle scraped spot, corrosion of aroused car parts, and engine tires are all sources of Zn and Cu in street dust (Bourliva et al. 2017). Lead also accumulates on street

dust from various sources, including Pb-containing gasoline, alloys, coating materials, pigment, paint, and other Pb-containing construction materials (Wang et al. 2016). Tannery, textiles, plastics, rubber, tire wear, paints, glass, and ceramics all contain chromium and cadmium (Mathur et al. 2016). In general, groundwater contamination, arsenical manures and pesticides, chromated copper arsenate treatment of wood, and coal use in power plants (Habib and Khan 2021; Habib et al. 2020; Habib et al. 2019a, 2019b), brick field escapes, and biomass burning are generally considered the dominant contributors of As in soils (Proshad et al. 2019; Kormoker et al. 2019; Fu et al. 2014). Moreover, mechanical waste and origin from nearby soil are viewed as the preeminent reasons for Ni in metropolitan street tidies (Delibašić et al. 2020). As a result, street dusts are rationally intricate, and their structures are not always consistent, which may be attributable to temporal shifts and the relatively brief retention time throughout the environment.

Toxic elements pose a risk to city dwellers due to their prevalence, consistency, and exposure to the environment, as they may reach the human body through inhalation, ingestion, and/or direct dermal touch (Rahman et al. 2019; Kabir et al. 2020b). Oral ingestion is by far the greatest harmful route for humans to be subjected to dust particles, since it occurs unintentionally with food and drink, and respecting children's hand-to-mouth practices (Li et al. 2017). Short-term metal exposure can cause breathing problems and skin irritation, whereas long-term metal exposure can lead to lung and skin cancer and/or kidney damage (Jayarathne et al. 2018). In this way, street dust testing is of particular concern because people breathe in residue going by every day or live in metropolitan areas with extremely hectic highways, and street dust is a medium by which potentially toxic elements can find their way into surface and underground soils and water, affecting people, creatures, and even vegetation (Razanica et al. 2014). Recently, some studies have been reported wherein the authors measured the risk of possible toxic compounds on human health (Rahman et al. 2019; Delibašić et al. 2020; Trojanowska and Świetlik 2019; Suryawanshi et al. 2016; Idris et al. 2018; Han et al. 2013). The USEPA health risk model has long been utilized to measure the health risks of radioactive metals in metropolitan street dust (USEPA 1989).

Gazipur is located in the central portion of Bangladesh, peripheral region of Capital Dhaka city, and prominently most important region of the country in terms of industry and communication. Moreover, urban areas of Gazipur are densely populated and most of the inhabitants are the low-income earning peoples. The content of urban street dust in this area is heavily affected by industrial pollution and traffic congestion. Power stations, manufacturing facilities, biomass burning, industrial metallurgical processes, contaminated water, and effluent released are all important sources of environmental pollution throughout the study region (Zakir et al. 2015).

The innumerable cars, improper vehicle servicing, inadequate stringent regulations for traffic management, low-quality exhaust emission monitoring during yearly vehicle inspections, lack of parking spaces and low-grade fuels, lack of more effective traffic regulations on existing highways, and an inappropriate mass transportation network are all factors that contribute to emissions from traffic sources. Various environmental pollution in Gazipur urban areas were conducted by numerous researchers, for instance, soil pollution assessment (Hossain et al. 2020; Zakir et al. 2015), surface water assessment (Ahmed et al. 2019; Rabbi et al. 2016), solid waste pollution (Baker et al. 2018; Islam et al. 2012), wastewater, and effluent assessment (Hossain et al. 2019; Shah et al. 2016). However, to date as we know, no comprehensive assessment has been done yet, regarding the street dusts of the urban areas of Gazipur. Consequently, very inadequate information regarding street dusts pollution is present of the study area. As a result, the analysis was performed to measure the concentrations of toxic compounds in street dust in Gazipur's urban areas, to assign a relationship between potentially toxic elemental abundances and their potential sources, and to assess the potential health risks of children and adults due to the exposure to potentially toxic elements in street dust via various exposure pathways. Furthermore, the concentrations of potentially harmful components in street dust of Gazipur's metropolitan areas were compared to those in other places around the globe.

## Experimental

### Study area

Gazipur is a major industrial zone of Bangladesh along with over-crowded populations and heavy traffic load communication systems (Hossain et al. 2020). With an area of 1741.53 km<sup>2</sup>, Gazipur is adjoined on the north by Mymensingh and Kishoreganj districts, on the south by Dhaka, Narayanganj, and Narsingdi districts, on the east by Narsingdi districts, and on the west by Dhaka and Tangail districts (BBS 2011). The Bangshi, Old Brahmaputra, Turag, Shitalakshya, and Balu are the major rivers that flow through the district (Ahmed 2012), with a varying temperature of 12.7 to 36.0 °C, and 2376 mm average annual rainfall (Zakir et al. 2015). Urban areas of Gazipur district are highly dominated by industrial and commercial activities, and inter-city communication arrangement. Metals factories, garment and staining, drug companies, cosmetics industry, machinery factory, fuel plant, publishing industry, ceramics factory, processing industry, brick fields, and so on, as well as about seven railway tracks and highways, are all present in this region (Hossain et al. 2020; Zakir et al. 2015). All these anthropogenic doings generate huge amounts of waste, wastewater, effluents and

atmospheric pollutants, and releases into the open environment. Consequently, these toxic pollutants and contaminants mixed up with various components of ecosystem and undergo different chemical and biochemical reactions (Kormoker et al. 2019). The sampling sites of the Gazipur urban area in Bangladesh are presented in Fig. 1.

### Sample collection and processing

Prior to the collection of street dusts samples, study areas were sensibly visited and investigated, after that sampling sites were identified based on the significance of the sites such as population density, traffic load, surroundings and land use pattern, presumed dust quality, and extent of pollution (Kabir et al. 2020b). Dusts samples were collected for metal analysis such as Zn, Pb, Cu, Cd, As, Cr, and Ni during dry season (February–March, 2019) by a pre-clean plastic dustpan and polyethylene brushes (Idris et al. 2018; Trojanowska and Świetlik 2019). From each sampling sites, four sub-dusts samples were collected and were mixed comprehensively to make a composite representative sample (Jiang et al. 2014). A total of 60 cumulative street dusts samples were collected from 15 sampling stations from Gazipur city corporation area. A widespread description of each sampling sites are presented in Table S1. Approximately 500 g street dusts samples were collected from each sampling sites by sweeping an area of ~ 2 m<sup>2</sup> including the impervious surface such as street, pavement, and gutter randomly (Delibašić et al. 2020; Trojanowska and Świetlik 2019). All of the samples were then stored in sealed polyethylene bags with appropriate labeling and finally transferred to the analytical laboratory. Collected samples were then air-dried until having constant weight and were sieved using 1 mm nylon sieve to get rid of coarse particle, stones, plastic, leaves, and other materials (Trojanowska and Świetlik 2019). At that point, the investigative samples were attained by grounding the samples with mortar and pestle, and were then sieved through 63 μm nylon sieve (Delibašić et al. 2020; Li et al. 2017). Finally, the samples were kept in Ziploc plastic-bags until analysis (Trojanowska and Świetlik 2019; Jehan et al. 2018).

### Sample analysis and quality assurance

All chemical analysis was done in an ultra-clean laboratory, using high purity chemicals and reagents (Khan et al. 2015a, 2015b). For preparation of solutions, Milli-Q water was used. Five percent HNO<sub>3</sub> was applied to clean and to soak the Teflon vessel and polypropylene containers for > 24 h followed by Milli-Q rinsing and drying. About 0.3–0.5 g of the dusts sample was treated by 6 ml conc. HNO<sub>3</sub> (69%) and 2 ml H<sub>2</sub>O<sub>2</sub> (30%) in a closed Teflon container and then digested through microwave digestion method. Digested sample was then shifted into Teflon beaker after filtration (syringe filter), and

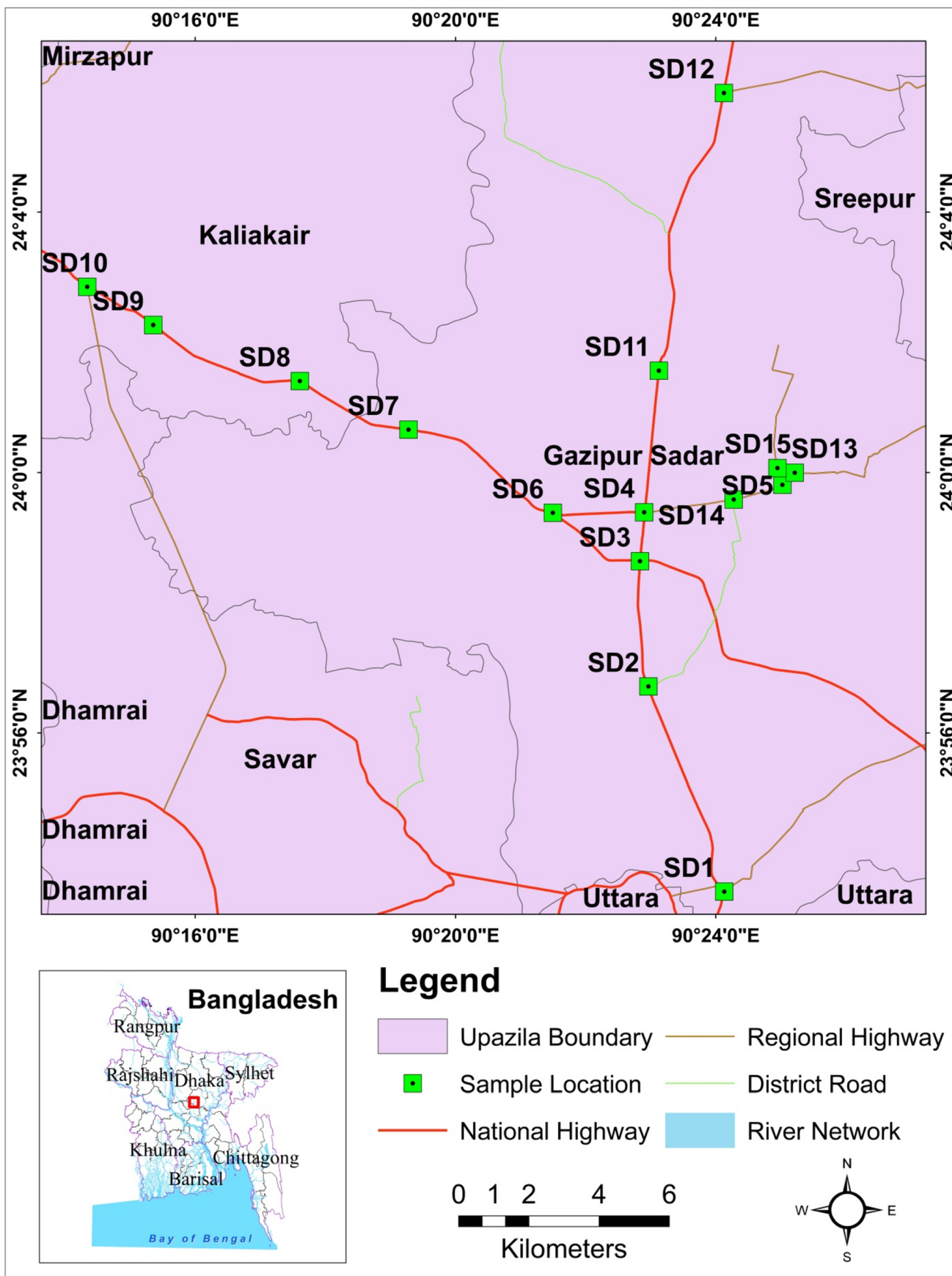


Fig. 1 Map showing the sampling sites of urban area of the Gazipur City, Bangladesh

made the volume up to 50 ml by Milli-Q water. Similar processes were conducted for the procedure blank. ICP-MS (Agilent 7700, CA) was used to determine the elemental contents. Data quality ensuring processes such as procedure blanks, duplicate sample measurements, repeated monitoring of reagent standards, and replicate measurements of reference materials (INCT-CF-3 from National Research Council of Canada) were same as those in Khan et al. (2015a, 2015b) and Islam et al. (2017a). Within the limit of analytical uncertainty, the studied elemental contents in the reference material were consistent with those in certificate values with 5–15% (RSDs in %,  $n = 3$ ) reproducibility. Potentially toxic elemental abundances in the street dusts were determined by following the standard method (Kormoker et al. 2019; Proshad et al. 2019; Islam et al. 2017b; Du et al. 2013).

### Environmental and ecological indices

The enrichment factor ( $EF$ ), geo-accumulation index ( $I_{geo}$ ), contamination factor ( $CF$ ), contamination degree ( $CD$ ), potential contamination ( $C_p$ ), modified degree of contamination ( $mCD$ ), Nemerow integrated pollution index ( $NIFI$ ), pollution load index (PLI), potential ecological risk (PER), and toxic units ( $TU_S$ ) indices were worked out to evaluate street dusts contamination extents. The salient features of these indices are described in Table 1.

### Human health risk assessment for urban street dust pollution

Health risk study consists of multiple steps (USDOE 2011; USEPA 1989) which include data measurements and assessment, exposure pathways appraisals, toxicity evaluation, and health risk estimation. For the assessment of non-carcinogenic risk (NCR) and carcinogenic risk (CR), seven metal(oid)s, i.e., Cu, Pb, Cr, Cd, Ni, Zn, and As, were considered in this study. Human exposure to hazardous elements through street dust can take places via three major pathways (Luo et al. 2012): (I) direct ingestion ( $CDI_{ingestion}$ ); (II) inhalation of re-suspended particulates emitted from dust through the nose and mouth ( $CDI_{inhalation}$ ); and (III) dermal absorption of hazardous elements in particles adhered to the exposed skin ( $CDI_{dermal}$ ). Both NCR and CR of these exposure pathways were evaluated in this study. In exposure estimation step, a specific characteristic approach for human exposure to street dust in the urban residential areas was applied, taking particularly care of the NCR exposure for both age group. Whereas CR was computed for the life-time exposure, assessed as the incremental potentiality of an individual, developing cancer over a life-time as a consequence of total exposure to the potential carcinogens. The received dose (in terms of chronic daily intake, CDI; i.e., average daily dose, ADD) through each three exposure pathways were evaluated and was estimated

using Eqs. (1)–(9) following USEPA (1989, 1997, 2001) and USDOE (2011). The definition of symbols, used values of Bangladesh-specific parameters and variables are presented in Tables (S2–S4).

Non-carcinogenic risk:

$$CDI_{ing-nc} = \frac{C_{soil} \times IngR \times EF \times ED}{BW \times AT} \times CF \quad (1)$$

$$CDI_{dermal-nc} = \frac{C_{soil} \times SA \times AF_{soil} \times ABS_d \times EF \times ED}{BW \times AT} \times CF \quad (2)$$

$$CDI_{inh-nc} = \frac{C_{soil} \times InhR \times EF \times ED}{PEF \times BW \times AT} \quad (3)$$

$$HQ = \frac{CDI_{nc}}{RfD} \quad (4)$$

$$HI = \frac{\sum HQ = HQ_{ingestion} + HQ_{dermal} + HQ_{inhalation}}{RfD_{ingestion}} + \frac{CDI_{dermal-nc}}{RfD_{ingestion} \times ABS_{GI}} + \frac{CDI_{inh-nc}}{RfC_{inhalation}} \quad (5)$$

Carcinogenic risk:

$$CDI_{ing-ca} = \frac{C_{soil} \times IngR \times EF \times ED}{BW \times AT} \times CF \times CSF_{ing} \quad (6)$$

$$CDI_{dermal-ca} = \frac{C_{soil} \times SA \times AF_{soil} \times ABS_d \times EF \times ED}{BW \times AT} \times CF \times CSF_{ing} \times ABS_{GI} \quad (7)$$

$$CDI_{inh-nc} = \frac{C_{soil} \times ET \times EF \times ED}{PEF \times 24 \times AT} \times IUR \times 10^3 \quad (8)$$

$$\text{Total risk} = \sum \text{Risk} = \text{Risk}_{ingestion} + \text{Risk}_{dermal} + \text{Risk}_{inhalation} \quad (9)$$

The chronic daily intake (CDI) for each metal(oid) and exposure rout ( $CDI_{ingestion}$ ,  $CDI_{dermal}$ , and  $CDI_{inhalation}$ ) were successively normalized by the affiliated reference dose to calculate hazard quotients [HQ or NCR; Eq. (4)]. For carcinogens, the doses for Pb, Ni, and As were multiplied by the corresponding slope factor (SF, Table 4) to obtain a level of excess life-time cancer risk [Eqs. (6 and 7)]. Although the effects from interactions between some metal(oid)s may take places in a synergistic manner (Luo et al. 2012; Xu et al. 2013), it was assumed that all the element-specific risks were additive, which allows to compute the cumulative NCR in terms of hazard index [HI, Eq. (5)], and NCR represented as the total cancer risk (Eq. (9)).

Reference toxicity values for dermal absorption were computed following the RAIS (Risk Assessment Information System) (USDOE 2011). Oral reference doses were multiplied and slope factors divided by a gastrointestinal absorption factor to yield the corresponding dermal values. The reference dose ( $RfD$ ) (mg/kg/day) is an estimation of maximum permissible risk on human population through daily exposure, considering the sensitive age group (children). In general, there

**Table 1** The contamination indices and ecological risk assessment indices used in this study

Index and formula	Description	Salient features	Limits/standards	References
Geo-accumulation index ( $I_{geo}$ ) $I_{geo} = \log_2 (C_n / 1.5 \times B_n)$	$I_{geo}$ , broadly used for assessment of metal contamination in soils by comparing the measured concentrations to pre-industrial background concentrations.	$C_n$ is the measured concentration of examined metal (n); $B_n$ is the geochemical background concentration of that corresponding metal (n); For calculation study used background value from CNEMC, because Bangladesh has no available background value for neighborhood country except China; the pre-industrial background values for Pb, Cu, Cr, Cd, Zn, Ni, and As in soils were 21.4, 21.4, 62.5, 0.11, 69.4, 28.9, and 11.09 mg/kg, respectively; the factor 1.5 is applied for the probable deviations in background values because of lithological effect.	class 0 ( $I_{geo} \leq 0$ ): uncontaminated, class 1 ( $I_{geo} = 0-1$ ): uncontaminated to moderately contaminated, class 2 ( $I_{geo} = 1-2$ ): moderately contaminated, class 3 ( $I_{geo} = 2-3$ ): moderately to strongly contaminated, class 4 ( $I_{geo} = 3-4$ ): strongly contaminated, class 5 ( $I_{geo} = 4-5$ ): strongly to extremely contaminated, and class 6 ( $I_{geo} \geq 5$ ): extremely contaminated.	Muller (1969) CNEMC (1990) Pan et al. (2017) Rahman et al. (2019) Sheijany et al. (2020) Khan et al. (2019a, 2019b)
Enrichment factor (EF) $EF = \frac{(C_M/C_{Al})_{Sample}}{(C_M/C_{Al})_{Background}}$	EF, a significant tool in assessing the degree of anthropogenic heavy metal contamination.	$(C_M$ and $C_{Al})_{Sample}$ is the average concentration of the examined metal in dusts sample; $(C_M$ and $C_{Al})_{Background}$ is the background concentrations used as the reference element; Al is used as the reference element in this study due to its geochemistry.	$EF < 1$ (no enrichment), $EF < 3$ (minor enrichment), $EF = 3-5$ (moderate enrichment), $EF = 5-10$ (moderately severe enrichment), $EF = 10-25$ (severe enrichment), $EF = 25-50$ (very severe enrichment) and $EF > 50$ (extremely severe enrichment)	Chen et al. (1991) Aminiyar et al. (2018) Cheng et al. (2018) Idris et al. (2018) Kormoker et al. (2019) Khan et al. (2021)
Contamination factor (CF) & Contamination degree (CD) $CF = (C_m) / (B_m)$ $CD = \sum_{i=1}^n CF_i$	CF, evaluates surface soil layer contamination by calculating the ratio between of measured concentration of each metal in the street dusts and the background concentration of that metal.	$C_m$ is the measured concentration of heavy metal in the dusts; $B_m$ is the preindustrial background concentration of concerned metal; CD was computed by the sum of the seven toxic metals in the street dusts of the study area.	$CF < 1$ : low, $1 \leq CF < 3$ : moderate, $3 \leq CF < 6$ : considerable, and $CF \geq 6$ : very high. $CD < 6$ : low, $6 \leq CD < 12$ : moderate, $12 \leq CD < 24$ : considerable, and $CD \geq 24$ : very high.	Hakanson (1980) Tomlinson et al. (1980) Trojanowska and Świetlik (2019) Roy et al. (2019) Duong and Lee (2011)
Potential Contamination ( $C_p$ ) $C_p = (Metal_{max}) / (Metal_{background})$	$C_p$ , estimate the amount of metallic elements detectable from soil analysis.	$Metal_{max}$ is the maximum concentration of metal in soil; $Metal_{background}$ is the average value of the same metal in a background level.	$C_p < 1$ : Unpolluted, low level of pollution; $1 \leq C_p \leq 3$ : Moderate polluted $3 \leq C_p$ : Strong polluted.	Davaultier and Rognerud (2001) Fahmida and Islam (2018) Aprile and Bouvy (2008) Weissmannová and Pavlovský (2017)
Modified degree of contamination (mCD) $mCD = \frac{1}{n} \sum_{i=1}^n CF_i$	mCD, estimates the pollution range of soil by		$mCD < 1.5$ : Nil to very low; $1.5 \leq mCD < 2.0$ : low;	Hakanson (1980)

**Table 1** (continued)

Index and formula	Description	Salient features	Limits/standards	References
	toxic metals; improved and comprehensive form of <i>CD</i> , which suggest an extent of the overall contamination of surface layers in a sampling area.	<i>mCD</i> is the modified contamination degree; <i>n</i> is the total number of metal elements considered; <i>i</i> is the <i>i</i> th pollutant; <i>CF<sub>i</sub></i> is the contamination factor of <i>i</i> th toxic metals.	2.0 ≤ <i>mCD</i> < 4.0: moderate; 4.0 ≤ <i>mCD</i> < 8.0: high; 8.0 ≤ <i>mCD</i> < 16.0: very high; 16.0 ≤ <i>mCD</i> < 32.0: extremely; <i>mCD</i> > 32: ultra-high.	Fahmida and Islam (2018) Abraham and Parker (2008) Maanan et al. (2014) Nouri and ABDEM (2016)
Nemerow integrated pollution index ( <i>NIPi</i> ) $NIPi = \sqrt{\frac{(PI_{avg.})^2 + (PI_{max.})^2}{2}}$	<i>NIPi</i> , integrated method compatible with extreme value, which not only considers the role of single elements but also highlights the importance of element with the most serious contamination.	<i>PI<sub>avg.</sub></i> is the average of the PI of all considered metals; <i>PI<sub>max.</sub></i> is the maximum value.	<i>NIPi</i> ≤ 0.7: non-pollution; 0.7 < <i>NIPi</i> ≤ 1: warning line of pollution; 1 < <i>NIPi</i> ≤ 2: low level of pollution; 2 < <i>NIPi</i> ≤ 3: moderate level of pollution; <i>NIPi</i> > 3: high level of pollution.	Nemerow (1985) Yu et al. (2004) Yang et al. (2011) Cheng et al. (2014) Shifaw (2018)
Pollution load index ( <i>PLI</i> ) $PLI = \sqrt[n]{(CF_1 \times CF_2 \times CF_3 \times \dots \times CF_n)}$	<i>PLI</i> , evaluates mutual pollution weight at divergent locations through the dissimilar metals in soils and sediments, and provided an evaluation of the inclusive toxicity grade of the each single sampling locations.	<i>PLI</i> is assessed as the <i>n</i> th root of the product of the multiplications of the contents; <i>PLI</i> value of 0, 1 and above 1 means perfection, existence of only baseline levels of pollutants and progressive deterioration of site quality, respectively.	<i>PLI</i> > 1 specifies pollution exists, conversely, if <i>PLI</i> < 1 designates there are nonexistence metal pollution.	Tomlinson et al. (1980) Roy et al. (2019) Kormoker et al. (2019) Kabir et al. (2020b)
Potential ecological risk ( <i>PER</i> ) $C_f^i = C_m^i / C_n^i$ $E_r^i = C_f^i \times T_r^i$ $PER = \sum_{i=1}^n E_r^i$	<i>E<sub>r</sub><sup>i</sup></i> , indicates sensitivity of the biotic community to the toxic elements and exemplifies the <i>PER</i> caused by the overall contamination.	<i>E<sub>r</sub><sup>i</sup></i> is the potential ecological risk coefficient of a single metal; <i>C<sub>f</sub><sup>i</sup></i> is the accumulating coefficient of metal ( <i>i</i> ); <i>T<sub>r</sub><sup>i</sup></i> is the toxic-response factor of metal ( <i>i</i> ); <i>C<sub>m</sub><sup>i</sup></i> is the value of toxic metal concentration in the dust samples; <i>C<sub>n</sub><sup>i</sup></i> is the pre-industrial background values in soils; toxic-response factors for Pb, Cu, Cr, Cd, Zn, Ni, and As were considered 5, 5, 2, 30, 1, 4, and 10, respectively.	<i>E<sub>r</sub><sup>i</sup></i> < 40 or <i>PER</i> < 150: low, 40 ≤ <i>E<sub>r</sub><sup>i</sup></i> < 80 or 150 ≤ <i>PER</i> < 300: moderate, 80 ≤ <i>E<sub>r</sub><sup>i</sup></i> < 600 or 300 ≤ <i>PER</i> < 600: considerable and 160 ≤ <i>E<sub>r</sub><sup>i</sup></i> < 320 or 600 ≤ <i>PER</i> : very high ecological risk for the soils.	Hakanson (1980) Chen and Zhou (1992) Proshad et al. (2019) Wu et al. (2010) Guo et al. (2010) Tong et al. (2020) Al-Shidi et al. (2020) Pan et al. (2016) Khan et al. (2020)
Toxic units ( <i>TU<sub>S</sub></i> ) $TU = C_M / PEL$ $\sum TU_S = TU_{metal(1)} + TU_{metal(2)} + TU_{metal(3)} + \dots + TU_{metal(n)}$	<i>TU<sub>S</sub></i> , is referred to as the sum of toxic units of measured contaminants. <i>TU</i> , represents the concentration above which adverse effects are expected to occur frequently, and is defined as the ratio of the weighed concentration of toxic metals in street dust to probable effect level ( <i>PEL</i> ).	<i>C<sub>M</sub></i> is the measured concentration of heavy metal in dusts; <i>PEL</i> is the probable effect levels value of corresponding heavy metals; $\sum TU_S$ is the product of toxic units for heavy metals in urban street dusts samples; In this study, <i>PEL</i> values for Pb = 91.3, Cu = 197, Cr = 90, Cd = 3.53, Zn = 315, Ni = 36, As = 17 mg/kg.	If the sum of toxic units ( <i>TU<sub>S</sub></i> ) for all investigated dust samples is greater than 4, then the moderate to serious toxicity of toxic metals are happened.	MacDonald et al. (2000) Zheng et al. (2008) Bai et al. (2011) Zeb and Ping (2017) Proshad et al. (2019) Kormoker et al. (2019)

are two  $R_fD$  for two exposure routes:  $R_fD$  (mg/kg/day) for ingestion,  $R_fD$  (mg/kg/day) × gastrointestinal absorption factor ( $ABS_{GI}$ ) for dermal exposure. If the CDI is less than the  $R_fD$ ,  $HQ \leq 1$ , which represents no adverse health effect, while, if the CDI exceeds the  $R_fD$ ,  $HQ > 1$ , which invoke potential adverse health effects (USEPA 1989, 2001). Moreover, guidelines for health risk estimation for chemical mixtures assumed that “simultaneous sub-threshold exposures to several chemical entities may result in an adverse health effect” and “the magnitude of the adverse effect will be proportional to the sum of the ratios of the sub-threshold exposures to acceptable exposures” (USEPA 1986). Thus, element-specific HQs can be summed up to obtain hazard index (HI) which can estimate the composite risks for mix pollutants Eq. (5) (USEPA 1989). The guidelines also demonstrate that any single chemical entity with an exposure level larger than toxicity value will cause the HI to surpass the unity, for multiple chemical exposures the HI can also exceed unity even though no single chemical exposure exceeds the corresponding  $R_fD$ . Generally, the excess cancer risks less than  $10^{-6}$  (a probability of 1 chance in 1,000,000 of an individual developing cancer) are considered to be insignificant whereas cancer risks above  $10^{-4}$  are considered unacceptable by most international regulatory agencies (e.g., USEPA 1989) and risks lying between  $10^{-6}$  and  $10^{-4}$  are generally considered an acceptable limit, depending on the circumstances and situation of exposure (Hu et al. 2012; Islam et al. 2020). The value equal to  $10^{-6}$  is also considered the carcinogenic target risk by USEPA (2011).

## Results and discussion

### Potentially toxic elemental concentrations in street dust samples

Descriptive statistics of measured elements (Pb, Cu, Cr, Cd, Zn, Ni, and As) concentration in street dust samples collected from urban areas of Gazipur region in Bangladesh, along with their comparison in relation to background value (BV-China), and maximum permissible contents of potentially toxic elements for agricultural soils (PTE-MPC) are summarized in Table 2. Abundances of Pb, Cu, Cr, Cd, Zn, Ni, and As in urban street dusts were fluctuated from 23.1 to 62.2, 26.3 to 72.1, 55.2 to 122.8, 4.9 to 19.7, 189.5 to 289.7, 18.4 to 52.7, and 1.1 to 3.7 mg/kg, respectively. Mean concentrations of investigated elements in urban street dusts were decreased with the following descending order  $Zn > Cr > Cu > Pb > Ni > Cd > As$ , and results indicated that concentrations were comparatively heterogeneous all over the study area. The urban activities such as industrial emissions, heavy traffic load, varied land uses, municipal discharges, domestic garbage, population density, rapid urbanization doings, and urban runoff of Gazipur city area might be the foremost causes of wide ranges of metal concentrations input

(Kabir et al. 2020a; Ali et al. 2016). To introduce of toxic metal pollution causes variance of coefficient (VC) is a useful tool, and higher and lower VC values indicate metals effluence are triggered artificially and naturally, respectively (Li et al. 2008). The VC values of Pb, Cu, Cr, Cd, Zn, Ni, and As in urban street dusts samples were observed 33.3, 34.2, 22.8, 59.0, 14.5, 31.0, and 38.3%, respectively (Table 2), described that degrees of metal contaminations mostly throughout the areas were influenced by means of anthropogenic activities. A study conducted by Duan et al. (2017) stated that toxic metal concentrations (Pb: 62.38, Cu: 44.38, and Zn: 42.09%) were governed through the anthropogenic actions; conversely, Guo et al. (2012) exposed that As (0.26 %) concentration in street dusts was affected by natural sources. The soil invertebrate toxicity reference values (SI-TRV) are applied to appraise whether metal concentrations in dusts pose potential risks to soil living invertebrates (USEPA 1999). The results of this work revealed that contents of Cu, Cr, Zn, and As were prominently several times greater than respective SI-TRV values proposed by USEPA; however, the contents of Pb, Cd, and Ni were lower than the SI-TRV. Considering the wildlife risk caused by metal(oid)s in the street dusts of the studied area stated that dusts samples poses ecological hazard.

At present investigation, the mean Pb content  $40.9 \pm 13.6$  mg/kg surpassed the reference soil background level 21.4 mg/kg (CNEMC 1990), and showed just about two times higher (Table 2). Comparable Pb levels in urban street dusts were also found by other authors such as Ahmed and Ishiga (2006), Duan et al. (2017), Tang et al. (2013), Wei et al. (2009), Rasmussen et al. (2001), and Delibašić et al. (2020) (Table 3). Moreover, several times higher Pb contents were also reported by numerous researchers like as Lu et al. (2010), Ma et al. (2016), Suryawanshi et al. (2016), Duong and Lee (2011), Faiz et al. (2012), Varrica et al. (2003), Robertson et al. (2003), Idris et al. (2018), and Sutherland and Tolosa (2000) (Table 3). The occurrence of higher level of Pb in street dusts might be from motor vehicle emissions, previous usage and leak of leaded gasoline, electrical vehicles (battery operated), brick kilns, industrial raw materials and combustion of fossil fuels, commercial activities, construction doings (pigment, paint, and coating), brakes wear, congested traffic, and lead weight loss of the wheel due to traffic (Idris et al. 2018; Trojanowska and Świetlik 2019; Rahman et al. 2019; Duan et al. 2017). Conversely, lower Pb contents were also documented in some other studies, for example, Rahman et al. (2019), Victoria et al. (2014), and Hughes (2002) (Table 3). Present assessment found that Pb contents in urban street dusts were significantly lower than the corresponding PTE-MPC value (Table 2) recommended by CEPA (1995).

Results of the study revealed that mean Cu content ( $44.9 \pm 15.4$  mg/kg) in urban street dusts was considerably higher than that in background reference value prescribed by the CNEMC (1990). Similar results also obtained in the previous literature documented by Rahman et al. (2019), Ahmed and Ishiga (2006), Chatterjee and Banerjee (1999), Tang et al. (2013),



**Table 2** Concentrations (mg/kg) of toxic metal in street dust of the different sampling sites

Sample ID	Pb	Cu	Cr	Cd	Zn	Ni	As
SD 1	39.2	32.4	95.2	6.9	243.9	37.8	1.7
SD 2	48.7	47.2	88.3	7.3	253.6	29.8	1.2
SD 3	49.4	52.8	88.7	9.2	189.5	38.4	1.5
SD 4	33.2	28.9	75.7	5.8	221.3	22.1	2.6
SD 5	25.6	26.3	68.1	5.2	257.8	18.4	1.7
SD 6	56.3	65.2	93.9	18.1	283.1	52.7	2.1
SD 7	62.2	72.1	122.8	18.6	238.2	45.9	3.7
SD 8	58.7	69.4	109.7	19.7	275.9	42.1	2.8
SD 9	51.9	58.6	97.8	12.4	226.6	47.2	3.1
SD 10	23.8	28.4	72.5	5.3	192.5	23.6	1.4
SD 11	42.4	42.4	79.6	5.2	289.7	28.3	1.1
SD 12	45.9	38.6	81.4	6.4	282.5	33.8	2.7
SD 13	23.1	39.4	55.2	4.9	201.4	25.4	1.1
SD 14	28.5	39.3	62.1	6.7	193.8	34.2	1.9
SD 15	25.2	32.4	58.9	5.1	237.3	22.7	2.4
Mean	40.9	44.9	83.3	9.1	239.1	33.5	2.1
Median	42.4	39.4	81.4	6.7	238.2	33.8	1.9
Standard Deviation (SD)	13.6	15.4	19.0	5.4	34.7	10.4	0.8
Maximum ( $M_{max}$ )	62.2	72.1	122.8	19.7	289.7	52.7	3.7
Minimum ( $M_{min}$ )	23.1	26.3	55.2	4.9	189.5	18.4	1.1
Variance of Coefficient (VC)	33.3	34.2	22.8	59.0	14.5	31.0	38.3
BV-China CNEMC (1990)	21.4	21.4	62.5	0.11	69.4	28.9	11.09
PTE-MPC CEPA (1995)	300	100	200	0.3	250	50	20
SI-TRV USEPA (1999)	100	32	0.2	10	199	100	0.25

PTE-MPC = maximum permissible concentrations of potential toxic elements for agricultural soils of China; BV-China = background value of soil environment in China; SI-TRV = soil invertebrate toxicity reference values

Faiz et al. (2012), Rasmussen et al. (2001), and Idris et al. (2018) (Table 3). However, prominently, more than a few folds higher amounts of Cu were found in street dusts samples of various city around the world reported by Duan et al. (2017), Duong and Lee (2011), Shi et al. (2008), Wei et al. (2009), Ma et al. (2016), Lu et al. (2010), Kamani et al. (2017), Yeung et al. (2003), Han et al. (2013), Suryawanshi et al. (2016), Robertson et al. (2003), Charlesworth et al. (2003), Ambade and Litrupa (2012), Varrica et al. (2003), Sezgin et al. (2003), Sutherland and Tolosa (2000), and Trojanowska and Świetlik (2019) (Table 3), and the study stated the greater extents of Cu pollution throughout the globe. The frequent and intensive application of brakes, gasoline uses automobiles, congested traffic, commerce activity, heavy equipment workshops, mechanical abrasion of vehicles, metallic parts and car components, tire abrasion, existence of Cu in diesel and lubricants, denting car workshop, uncontrolled industrial and waste burning activities, industrial waste and effluents, large-scale automobile, and industrial emission are the notable sources of extensive level of Cu in street dust (Idris et al. 2018; Rakib et al. 2014; Rahman et al. 2019; Trojanowska and Świetlik 2019). On the other hand, lower concentration of Cu in urban street dusts were recorded by researchers, for instance,

Ferreira-Baptista and Miguel (2005), Victoria et al. (2014), and Hughes (2002) (Table 3). However, Cu concentrations in all of the dusts samples throughout the study area were much lower than the PTE-MPC value (Table 2) endorsed by CEPA (1995).

The mean Cr content ( $83.3 \pm 19.0$  mg/kg) was comparatively higher than the corresponding background reference value (62.5 mg/kg) (Table 2) indicated that study areas were might be heavily influenced by the industrial activities along with the geo-morphological settings. Analogous Cr concentrations in urban dust were also found when compared to other research conducted by Tang et al. (2013). Chromium presence in urban street dusts can be justified by both natural (concentration of Cr-bearing minerals) as well as by anthropogenic (industrial activities such as tanneries and textile mills) origins which are releasing Cr-based oxidants such as dichromate, chromate, etc. (Tamim et al. 2016; Proshad et al. 2019; Ahsan et al. 2019). Hence, the waste and effluents released from such industrial activities were mostly accountable for higher level of Cr in the exposed dusts sample. Besides, worldwide elevated level of Cr concentrations were available in previous literature accomplished by several scientist including Ahmed and Ishiga (2006), Ambade and Litrupa (2012),

**Table 3** Comparison of toxic metal (mg/kg) concentrations in street dust of Gazipur with major cities of the world

City	Pb	Cu	Cr	Cd	Zn	Ni	As	References
Gazipur, Bangladesh	40.9	44.9	83.3	9.1	239.1	33.5	2.1	Present Study
Dhaka, Bangladesh	47.0	46.8	104.3	–	121.3	27.0	4.0	Ahmed and Ishiga (2006)
Dhaka, Bangladesh	18.9	49.8	144.3	11.6	239.1	37.1	8.1	Rahman et al. (2019)
Guiyang, China	67.8	129.8	131.2	0.62	185.9	61.1	11.3	Duan et al. (2017)
Shanghai, China	294.9	196.8	159.3	1.23	733.8	83.9	–	Shi et al. (2008)
Beijing, China	50.40	64.23	77.45	0.47	–	23.70	–	Tang et al. (2013)
Suzhou, China	262.2	104.8	25.7	2.45	376.9	14.7	13.4	Ma et al. (2016)
Baoji, China	433.2	123.2	126.7	–	715.3	48.8	19.8	Lu et al. (2010)
Urumqi, China	53.53	94.54	54.28	1.17	294.47	43.28	–	Wei et al. (2009)
Ulsan, Korea	111.2	134.6	–	1.9	179.9	20.4	–	Duong and Lee (2011)
Eslamshahr, Iran	71.3	239.0	35.1	0.34	573.0	42.4	–	Kamani et al. (2017)
Kuala Lumpur, Malaysia	85.9	130.8	39.1	1.64	204.9	18.9	–	Han et al. (2013)
Istanbul, Turkey	211.9	208.5	–	2.3	520.8	33.3	–	Sezgin et al. (2003)
Bolgatanga, Ghana	5.5	7.5	6.3	0.19	3.4	2.1	0.15	Victoria et al. (2014)
Kolkata, India	536.0	44.0	54.0	3.12	159.0	42.0	–	Chatterjee and Banerjee (1999)
Delhi, India	120.7	191.7	148.8	2.65	284.5	36.4	–	Suryawanshi et al. (2016)
Bhopal, India	534.6	126.4	121.5	–	169.0	45.3	18.4	Ambade and Litrupa (2012)
Islamabad, Pakistan	104.0	52.0	–	5.0	116.0	23.0	–	Faiz et al. (2012)
Luanda, Angola	351.3	41.7	25.7	1.15	316.6	10.0	5.0	Ferreira-Baptista and Miguel (2005)
Ottawa, Canada	39.5	65.8	43.3	0.37	112.5	15.2	1.3	Rasmussen et al. (2001)
Japan	20.4	36.9	41.3	0.41	63.8	28.5	9.02	Hughes (2002)
Hong Kong	120.0	110.0	–	–	3840.0	28.60	–	Yeung et al. (2003)
Palmero, Italy	544.0	98.0	218.0	1.1	207.0	14.0	6.67	Varrica et al. (2003)
Hawaii, USA	106.0	167.0	273.0	–	434.0	177.0	–	Sutherland and Tolosa (2000)
London, England	265.0	113.0	–	–	653.0	-	–	Robertson et al. (2003)
Birmingham, England	106.0	466.9	–	–	534.0	41.1	–	Charlesworth et al. (2003)
Radom, Poland	88.0	239.0	53.5	–	618.0	50.0	–	Trojanowska and Świetlik (2019)
Bosnia and Herzegovina	52.49	30.01	33.17	3.18	81.72	73.00	–	Delibašić et al. (2020)
Khamees-Mushait, Saudi Arab	126.43	49.37	186.49	1.16	117.98	–	–	Idris et al. (2018)

Rahman et al. (2019), Shi et al. (2008), Duan et al. (2017), Lu et al. (2010), Suryawanshi et al. (2016), Sutherland and Tolosa (2000), Varrica et al. (2003), and Idris et al. (2018) (Table 3). It can be assumed that the major origins of Cr in the urban street dusts samples might be the cars corrosions, vehicular traffic, combustion of fossil fuels, power generation plants, tannery and dying industries, metal works, application of anti-corrosion paints, and vehicle scrap workshops (Rahman et al. 2019; Delibašić et al. 2020; Kormoker et al. 2019; Idris et al. 2018; Tan et al. 2018). In addition, according to Chen et al. (2014) for the production of stainless steel, automobile parts, aluminum alloy, and titanium alloy Cr was expansively used. Oppositely, lower level of Cr concentrations than background concentration were reported by numerous literature conducted by Kamani et al. (2017), Ferreira-Baptista and Miguel (2005), Han et al. (2013), Ma et al.

(2016), Victoria et al. (2014), Chatterjee and Banerjee (1999), Hughes (2002), Rasmussen et al. (2001), Trojanowska and Świetlik (2019), and Delibašić et al. (2020) (Table 3). On the other hand, Cr concentrations were found to be much below the PTE-MPC value (Table 2) recognized by CEPA (1995) for urban dust samples of the studied area.

Mean abundance of Ni was (33.5 ± 10.4 mg/kg) marginally higher and in some extent only one time exceeded the background reference value (28.9 mg/kg) (Table 2). Consequently, reasonably identical Ni contents were also found in various studies conveyed by Rahman et al. (2019), Lu et al. (2010), Ambade and Litrupa (2012), Wei et al. (2009), Kamani et al. (2017), Charlesworth et al. (2003), Chatterjee and Banerjee (1999), and Suryawanshi et al. 2016 (Table 3). Nevertheless, Ni concentrations were elevated and pointedly higher than the

present study and background reference level in urban dust samples of many studies, for example, Duan et al. (2017), Shi et al. (2008), Sutherland and Tolosa (2000), Trojanowska and Świetlik (2019), and Delibašić et al. (2020) (Table 3). Although, the motor oils contain trace levels of Ni, however, combustion of diesel-fuel is the core cause of Ni existence in urban street dust (Delibašić et al. 2020). As well, corrosion of body and parts in cars, accidental spillages of Ni containing materials, municipal and industrial waste, and lithogenic factors (origin from local soil) could be the major sources of Ni in urban terrestrial environment (Trojanowska and Świetlik 2019; Kormoker et al. 2019; Duan et al. 2017). Nickel concentrations in urban dusts were considerably lower in different former studies (Duong and Lee 2011; Ahmed and Ishiga 2006; Ma et al. 2016; Tang et al. 2013; Han et al. 2013; Ferreira-Baptista and Miguel 2005; Victoria et al. 2014; Faiz et al. 2012; Hughes 2002; Varrica et al. 2003; Yeung et al. 2003; Rasmussen et al. 2001) than this study (Table 3). Furthermore, all of the urban street dust samples showed lower amount of Ni contents when compared to PTE-MPC value (Table 2) documented by CEPA (1995).

In this study, the mean abundance of Cd in urban street dusts sample was observed  $9.1 \pm 5.4$  mg/kg (Table 2). In contrast, the reference background value for Cd in soil is 0.11 mg/kg (CNEMC 1990). Results of the study revealed that eminent Cd contents in street dust samples were almost 88-fold greater than that in background level. These elevated Cd concentrations can be originated from industrial activity, Cd-plated items, metal processing, and atmospheric emission, etc. (Proshad et al. 2019). Moreover, Cd, Pb, and Zn are the prime component of battery, and continuously battery-operated electrical vehicles which are increasingly used in Gazipur urban areas. But, these used batteries are not appropriately collected for reprocess, and consequently, the used batteries are discarded in the ambient environment (Rahman et al. 2019). Cadmium contents in dusts of this analysis were compared to those of other exploration performed in Bangladesh as well as in other countries and reveal that Cd abundance in this investigation was just about comparable to Rahman et al. (2019), and higher than those found in other studies (Duong and Lee 2011; Duan et al. 2017; Ma et al. 2016; Chatterjee and Banerjee 1999; Wei et al. 2009; Kamani et al. 2017; Shi et al. 2008; Han et al. 2013; Tang et al. 2013; Sezgin et al. 2003; Victoria et al. 2014; Suryawanshi et al. 2016; Ferreira-Baptista and Miguel 2005; Faiz et al. 2012; Rasmussen et al. 2001; Hughes 2002; Varrica et al. 2003; Idris et al. 2018; Delibašić et al. 2020 (Table 3). Besides, urban street dust samples of the studied areas were exhibited significantly higher amount of Cd contents with respect to PTE-MPC value (Table 2) renowned by CEPA (1995). In addition, Cd is successively associated with deterioration of tyres and engine parts (Duan et al. 2017), in the same way, Cd exists as alloys in accumulators of motor vehicles or in carburetors (Delibašić et al. 2020).

The mean As concentration in urban street dusts were perceived  $2.1 \pm 0.8$  mg/kg, and the background reference value for this element is 11.09 mg/kg (Table 2). Study revealed that As concentrations in all of the street dust samples of the investigated areas were prominently lower than the respective elemental background level. Generally, As in urban street dusts can be originated from both anthropogenic and natural sources, for instance, treatment of wood with chromated copper arsenate, use of arsenic-contaminated groundwater, burning of coal in power stations, and application of arsenical fertilizers and pesticides (Proshad et al. 2019; Fu et al. 2014). Furthermore, emission and waste from incineration activities and brick fields in urban region may contribute to the higher abundances of As in urban soils (Kormoker et al. 2019). Average abundance of As in the street dust samples of this study was found to be considerably lower than the numerous previous studies conducted in Bangladesh as well as in other countries, for example, Ferreira-Baptista and Miguel (2005), Ahmed and Ishiga (2006), Duan et al. (2017), Ambade and Litrupa (2012), Ma et al. (2016), Hughes (2002), Lu et al. (2010), Rahman et al. (2019), and Varrica et al. (2003) (Table 3). However, these As level might be associated to both anthropogenic (due to industrial activities) as well as natural (geogenic influences) sources (Rahman et al. 2019). The As concentration in urban street dusts of this study was compared to CEPA (1995) standard quality guidelines and it was cleared that As content level was significantly lower than PTE-MPC (Table 2).

In present assessment, among the all studied heavy metal(oid)s, Zn contents was high extended from 189.5 to 289.7 mg/kg (Table 2). The mean Zn content was found to be  $239.1 \pm 34.7$  mg/kg for all of the urban street dust samples that surpassed the reference soil background level (69.4 mg/kg) almost by 4-folds. A more or less comparable Zn contents in the fine portion of urban street-dust were obtained as well by other researchers, for instance, Wei et al. (2009), Rahman et al. (2019), Han et al. (2013), Varrica et al. (2003), and (Suryawanshi et al. (2016) (Table 3). It has been suggested that these Zn concentrations could be originating from vehicle traffic may be from diesel exhaust emissions, tire wears, and brakes wear (Trojanowska and Świetlik 2019).

Zn concentration of current examination was compared with that in PTE-MPC (CEPA 1995) soil standard quality guidelines and revealed that Zn concentration was higher at seven sampling sites, and lower at rest of the sampling sites (Table 2). Zinc concentration of this study was compared with those studies executed in Bangladesh as well as in other countries, and revealed that elevated concentrations of Zn, counting, up to 2 times higher than the present investigation (Shi et al. 2008; Ferreira-Baptista and Miguel 2005; Ma et al. 2016; Lu et al. 2010; Yeung

et al. 2003; Kamani et al. 2017; Robertson et al. 2003; Sezgin et al. 2003; Ferreira-Sutherland and Tolosa 2000; Charlesworth et al. 2003; Trojanowska and Świetlik 2019) (Table 3). Results of the study recommended that Zn might have entered into urban street dust owing to anthropogenic activities particularly from metal smelting and motor-vehicle industries (Rahman et al. 2019). However, Zn is a common element in car tires, lubricants, and carburetors (Al-Khashman 2004). Moreover, ZnO is mostly used in tire tread rubber and lesser amounts of organo-zinc compounds are also used for rubber vulcanization (Rahman et al. 2019). Consequently, it can be suggested that the pollution of Zn in street dust predominantly occurs from heavy traffics, heavy equipment workshops, denting vehicle workshops, mechanical abrasion of vehicles, and metal smelting workshops (Idris et al. 2018; Delibašić et al. 2020; Al-Khashman 2004). Besides, lower level of Zn contents were also documented by numerous previous studies (Rasmussen et al. 2001; Ahmed and Ishiga 2006; Duan et al. 2017; Delibašić et al. 2020; Victoria et al. 2014; Chatterjee and Banerjee 1999; Ambade and Litrupa 2012; Faiz et al. 2012; Hughes 2002; Duong and Lee 2011; Yeung et al. 2003; Idris et al. 2018) than the present exploration (Table 3).

### Spatial distributions of toxic metal(oid)s in urban street dusts

Elemental distributions throughout the study area and probable source identification could be conceivable through GIS application (Guo et al. 2012). The extents of potentially toxic metal(oid)s' pollution for both rural and urban areas can be portrayed effectively through spatial distribution map parenthetically. In this study, Kriging method of spatial analyst tools was used for interpolation of heavy metal distribution, and ArcGIS 10.4 (Version 10.4, ESRI, USA) was used in this purpose.

Spatial distributions of Cu, Pb, Cr, Zn, Cd, Ni, and As in street dust samples of Gazipur urban areas are displayed in Fig. 2. The topography of the study area is uniform, consequently, concentration of potential toxic elements in dusts varies based on surrounding factors, such as, traffic volume, population density, street setting alike street junction, main street, bus stoppage, and land use pattern (residential, commercial, or industrial zone). The study demonstrated that the concentration of Pb, Cu, and Cr were found high at the sampling sites SD7 (Konabari) and SD8 (Mouchak), might be due to the higher numbers of traffic, dense population, commercial doings, and heavy industrial activities in those areas are mainly accountable for higher concentrations (Rahman et al. 2019; Guo et al. 2012). Massive use of electrical vehicles (battery operated), brick kilns operation, construction deeds, pre-used leaded gasoline, industrial raw materials, and combustion of fossil fuels that derived from industrial exhausts could be the reasons for

accumulating high concentration of Pb in the study sites. However, higher Cr concentrations may be owing to the increased number of large vehicles, dying factories, and others gasoline uses as well over time. Study area are highly influenced gasoline uses automobiles, congested traffic, mechanical abrasion of vehicles, tire abrasion, existence of Cu in diesel and lubricants, denting car workshop, large-scale industrial emission, those are responsible for higher concentration of Cu in street dust. However, spatial distribution of Cd depicted that higher values were found at the sampling sites of SD8 (Mouchak), SD7 (Konabari), and SD6 (Nawjor), respectively, might be because of rapid growing heavy industries where Cd has been using for several purposes like painting, metal plating and so on. It has been found that the higher level of Zn in street dusts were found 289.7, 283.1, and 282.5 mg/kg, respectively at the sampling sites of SD11 (Salna), SD6 (Nawjor), and SD12 (Rajendrapur), respectively. However, most of the sampling sites are representing higher concentration of Zn comparing to the soil background value of Zn (69.9 mg/kg) (CNEMC 1990). Generally, Zn mixtures have been used in lubricating oils, thus, Zn content mixed in urban soils by wear and tear of tires. Besides, long-term unused vehicle parts that mixes with soil over time, emission from vehicles, and industrial emissions result in accretion of Zn in urban soil. The result of the study showed that both sampling sites of SD7 and SD9 have higher concentrations of Ni and As. Particularly, the highest value of Ni and As has been found in the sampling location of SD6 and SD7, respectively. Ni concentrations found in this study for most of the sampling areas are significantly higher than soil back ground level of Ni (28.9 mg/kg) by China Environmental Centre (CNEMC 1990). On the other hand, concentrations of As for all sites were found relatively lower than other heavy metals in this study. Heavy traffic and population density, metal smelting workshops, and car repairing workshops are mainly considered for both Ni and As accumulation in dust throughout the study area. In addition, various anthropogenic activities such as burning fossil fuel, improper waste dumping, smelting of metal, and making and abrasion of car parts are also responsible for Ni contamination in the urban areas of Gazipur.

Based on the assessment, it can be said that the street dust of the sampling sites of SD6, SD7, SD8, and SD9 are highly contaminated by studied toxic metals. The causes behind those toxic metals (Pb, Cu, Cr, Cd, As, Ni, and Zn) contamination in street dust of Gazipur urban area are mainly associated with multiple aspects such as condition of traffic, land use, industrial patterns, geomorphology, population load, street types, and so on. Moreover, study stated that more

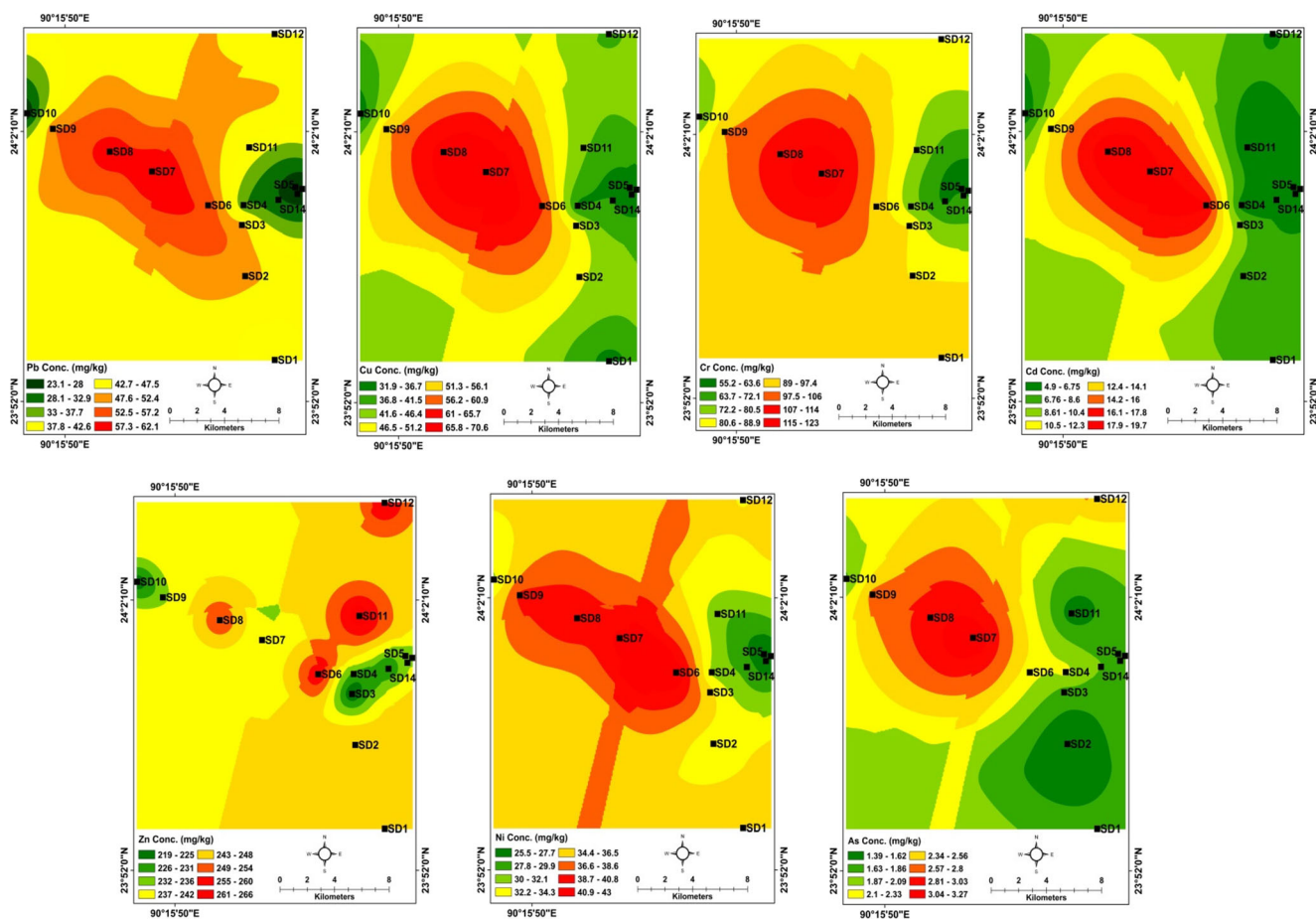


Fig. 2 Spatial distribution of toxic metal concentrations in urban street dusts of Gazipur City, Bangladesh

anthropogenic activities showed a considerable increase in toxic metal contaminations of street dust in the study sites.

**Deciphering interrelationships and possible sources of potential toxic elements in street dust**

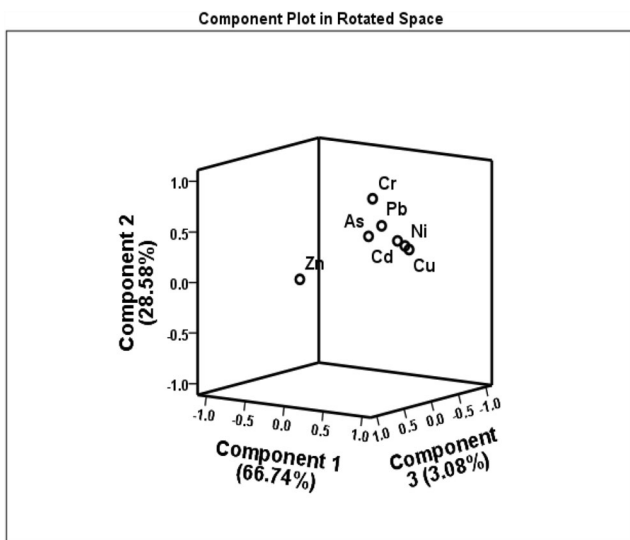
To decipher the interrelations between the studied metals and to identify their possible sources in the street dusts, Pearson’s correlation coefficient analysis (CCA), principal component analysis (PCA), and cluster analysis (CA) were performed as also widely used in other related studies (Jose and Srimuruganandam 2020; Suryawanshi et al. 2016; Jiang et al. 2017; Khan et al. 2019a, 2019b; Zglobicki et al. 2018; Ahmed et al. 2021). The results of CCA revealed significant positive correlations among most of the studied metals (Table 4). Lead, Cu, Cr, Ni, and Cd showed significantly strong positive correlations ( $r = 0.761–0.915$ , at  $p < 0.01$ ) with each other, indicating that these toxic metals are originated from the common anthropogenic pollution sources in the studied area. In street dusts, these toxic metals (Pb, Cu, Ni, Cr, and Cd) are found to be originated by the release of particles as a result of industrial and vehicular emissions. Similar findings were also reported by Suryawanshi et al. (2016) in Delhi,

India. On the other hand, As showed significant ( $p < 0.05$ ) positive correlation with Cd (0.616) and Cr ( $r = 0.577$ ), indicating that the source of As is possibly the industrial activities and not the vehicular emissions. However, no significant relationship was found between Zn and other metals although weak correlation ( $r = 0.462$ ) was observed with Pb, pointing out the traffic activities as the probable source of Zn.

Table 4 Pearson correlation coefficients among toxic metals in street dust samples

Metals	Pb	Cu	Cr	Cd	Zn	Ni	As
Pb	1						
Cu	.889**	1					
Cr	.915**	.791**	1				
Cd	.834**	.927**	.823**	1			
Zn	.462	.264	.350	.312	1		
Ni	.838**	.859**	.761**	.837**	.224	1	
As	.514	.501	.577*	.616*	.168	.484	1

\*\* Correlation is significant at the 0.01 level (two-tailed). \* Correlation is significant at the 0.05 level (two-tailed)



**Fig. 3** Principal component analysis (PCA) of toxic metals in street dust samples

The results of PCA exhibited that the studied toxic metals are well represented by three principal components (PCs) which can explain 98.40% of total variance (Fig. 3). The first PC (PC1) was dominated by four metals (Pb, Cu, Ni, and Cd) and explained 66.74% of total variance. PC2 was governed by two metals (Cr and As) and accounted for 28.58% of the total variance, whereas PC3 was dominated by Zn and explained only 3.08% of the total variance. The results of PCA were found to be in line with the results obtained by CCA which revealed that the toxic metals in the street dusts are possibly of anthropogenic origins. The PC1 could explicitly describe the influence of both industrial and heavy traffic-related activities on the toxic elemental contamination in street dust samples. Sources of Cu, Pb, Cd, and Ni may include fuel consumption, leakage of oil and lubricants, wear and tear of engines and other vehicular components, and exhaust emissions (Roy et al. 2019; Zglobicki et al. 2018; Jiang et al. 2017). Moreover, Cu and Ni can be released directly or indirectly from the metal processing, smelting, and electronic components manufacturing industries (Zhao et al. 2019; Jiang et al. 2017; Suryawanshi et al. 2016), while paint flakes can contribute to Pb and Cd pollution in the street dust (Zhao et al. 2019). The PC2 could explain the influence of industrial activities on Cr and As contamination in the street dust. Cr is an important component of many metal alloys, and thus, can be originated from the metal processing industries in the study area (Suryawanshi et al. 2016). However, yellow road marking on the roads can be another possible source of Cr contamination (Murakam et al. 2007), which cannot be underestimated. On the other hand, As was also reported to be released from coal combustion activities as well as industrial emissions (Xu et al. 2013). The PC3 could describe the possible sources of Zn which can be released as a result of vehicular activities like use of brake pads, tear of tires, and galvanization operations (Suryawanshi et al. 2016, Jose and Srimuruganandam 2020).

The results of CA are presented in Fig. S1 that grouped both the toxic metals and sampling sites into three major clusters. In case of toxic metals, the cluster 1 was formed with Pb, Cu, Cd, Cr, and Ni, while the clusters 2 and 3 were formed with As and Zn, respectively (Fig. S1a), indicating nearly common origins and accumulation pathways of the metals in the street dust. On the other hand, while considering the sampling sites, the cluster 1 was constructed with five sampling sites (1, 2, 4, 8, and 10), the cluster 2 was formed with seven sampling sites (5, 6, 11, 12, 13, 14, and 15), and the cluster 3 was developed with three sampling sites (3, 7, and 9) (Fig. S1b), representing similar characteristic features including level of pollution, extent of street traffic activities, and industrial settings.

### Environmental pollution and ecological risk assessment

Geo-accumulation index was used in this analysis to determine the levels of contamination of potentially toxic elements in the samples of street dust. Table S5 summarizes the findings of  $I_{geo}$ . The  $I_{geo}$  value at fifteen sites in Gazipur's urban areas were mostly in the following manner: Cd ( $5.60 \pm 0.73$ ) > Zn ( $1.19 \pm 0.21$ ) > Cu ( $0.41 \pm 0.48$ ) > Pb ( $0.27 \pm 0.51$ ) > Cr ( $-0.20 \pm 0.33$ ) > Ni ( $-0.44 \pm 0.46$ ) > As ( $-3.11 \pm 0.56$ ). At all sites, the  $I_{geo}$  value including Cr, Ni, and As were in the uncontaminated group, specifying that Cr, Ni, and As might have natural origins in the street dusts including the geological factors of the areas (Sheijany et al. 2020). The value of  $I_{geo}$  for Pb and Cu, at almost all sites, were found to be uncontaminated to moderately contaminated; on the other hand, Zn and Cd were prominently exposed moderately contaminated and extremely contaminated classes at more or less all examined sites, respectively. Furthermore, comparable  $I_{geo}$  value for Cu, Pb, and Zn in Khamees-Mushait (Saudi Arabia) city street dusts ranged from uncontaminated to moderately polluted, possibly due to road congestion, trade operations, and workshops for vehicle and equipment servicing (Idris et al. 2018). Higher  $I_{geo}$  values for Cd in the study area can be linked to industrial doings (Proshad et al. 2019), as well as with the higher load of electrical vehicles (operated by battery) (Rahman et al. 2019). In the street dusts of Radom, Poland, Trojanowska and Świetlik (2019) discovered a parallel pattern for the  $I_{geo}$  of Ni, Pb, Cu, and Zn. Moreover, Kamani et al. (2017) ended an analogous statement in street dusts of Eslamshahr, Iran.

In this assessment, the mean of EF was highest for Cd ( $46.54 \pm 27.45$ ), and declined in the following order of Zn ( $1.93 \pm 0.28$ ) > Cu ( $1.18 \pm 0.40$ ) > Pb ( $1.07 \pm 0.36$ ) > Cr ( $0.75 \pm 0.17$ ) > Ni ( $0.65 \pm 0.20$ ) > As ( $0.10 \pm 0.04$ ) (Table S6). The mean EF values for Cr, Ni, and As belongs to category 1 or no enriched metals; for Pb, Cu, and Zn goes to category 2 or minor enriched metals. Similarly, the metals Zn, Pb, and Cu were moderately enriched in street dusts in Khamees-Mushait (Saudi Arabia), possibly because of the existence of denting vehicle workshops, heavy

traffic, the pesticides and fertilizers industry, and scrap stores and vehicle workshops (Idris et al. 2018). Conversely, the mean EF values for Cd belongs to category 6 or very severe enriched metal. The results of the study revealed that all of the street dusts were considerably contaminated by Cd in the investigated sites. Results of this study again point out the potential impacts from industry and battery operated vehicles in the study areas (Rahman et al. 2019; Proshad et al. 2019). Aminiyan et al. (2018) documented moderate to high enrichment for Zn, Cu, Ni, As, Cd, Pb, and Cr in the street dusts of Rafsanjan city of Iran that is partially related to our results. Different results of dusts EF could be caused by natural variation of soil in that area, in conjunction with the climate of the region and the surrounding vegetation (Shejany et al. 2020; Liu et al. 2013).

The Pb, Cu, Cr, Cd, Zn, Ni, and As had mean CF values of  $1.91 \pm 0.64$ ,  $2.10 \pm 0.72$ ,  $1.33 \pm 0.30$ ,  $82.91 \pm 48.90$ ,  $3.45 \pm 0.50$ ,  $1.16 \pm 0.36$ , and  $0.19 \pm 0.07$ , respectively, in the current analysis, and the decreased in the order of  $Cd > Zn > Cu > Pb > Cr > Ni > As$ . (Table 5). For all street dust samples collected in the studied area, the extent of Cd contamination was extremely high, whereas Zn showed the considerable grade of contamination. These may be as a result of massive industrial launches as well as metal operations (Kormoker et al. 2019).

**Table 5** Contamination factor (CF), contamination degree (CD), and pollution load index (PLI) values of toxic metals for street dust of urban areas of Gazipur

Sites	Contamination factor (CF)							CD	PLI
	Pb	Cu	Cr	Cd	Zn	Ni	As		
SD 1	1.83	1.51	1.52	62.73	3.51	1.31	0.15	72.57	2.11
SD 2	2.28	2.21	1.41	66.36	3.65	1.03	0.11	77.05	2.12
SD 3	2.31	2.47	1.42	83.64	2.73	1.33	0.14	94.03	2.29
SD 4	1.55	1.35	1.21	52.73	3.19	0.76	0.23	61.03	1.86
SD 5	1.20	1.23	1.09	47.27	3.71	0.64	0.15	55.29	1.61
SD 6	2.63	3.05	1.50	164.55	4.08	1.82	0.19	177.82	3.11
SD 7	2.91	3.37	1.96	169.09	3.43	1.59	0.33	182.69	3.46
SD 8	2.74	3.24	1.76	179.09	3.98	1.46	0.25	192.52	3.28
SD 9	2.43	2.74	1.56	112.73	3.27	1.63	0.28	124.63	2.91
SD 10	1.11	1.33	1.16	48.18	2.77	0.82	0.13	55.50	1.57
SD 11	1.98	1.98	1.27	47.27	4.17	0.98	0.10	57.76	1.92
SD 12	2.14	1.80	1.30	58.18	4.07	1.17	0.24	68.92	2.30
SD 13	1.08	1.84	0.88	44.55	2.90	0.88	0.10	52.23	1.53
SD 14	1.33	1.84	0.99	60.91	2.79	1.18	0.17	69.22	1.88
SD 15	1.18	1.51	0.94	46.36	3.42	0.79	0.22	54.42	1.72
Minimum	1.08	1.23	0.88	44.55	2.73	0.64	0.10	52.23	1.53
Maximum	2.91	3.37	1.96	179.09	4.17	1.82	0.33	192.52	3.46
Mean	1.91	2.10	1.33	82.91	3.45	1.16	0.19	93.04	2.24
SD ( $\pm$ )	0.64	0.72	0.30	48.90	0.50	0.36	0.07	50.85	0.64
Median	1.98	1.84	1.30	60.91	3.43	1.17	0.17	69.22	2.11

However, Pb, Cu, Cr, and Ni in street dusts revealed moderate grade, on the contrary, As pretended lower grade of contamination. In street dust samples from Cairo (Abdel-Latif and Saleh 2012) and Delhi (Suryawanshi et al. 2016) obtained similar Zn and Pb grades. Besides, almost alike results also attained for the Cu and Ni (Roy et al. 2019). In contrast, Trojanowska and Świetlik (2019) observed very high CF values of Cu, Zn, and Pb in samples of street dusts of Radom, Poland. The study's findings revealed that CD values of all metals across all sampling locations suggested an aggregate contamination caused by the combined effect of all toxic metals examined (Rahman et al. 2019; Roy et al. 2019). Mean CD value was  $93.04 \pm 50.85$ , that is grouped into the very high ( $CD \geq 24$ ) magnitudes of pollution (Table 5). The outcomes of this study is more or less similar to the earlier studies reported by Abdel-Latif and Saleh (2012), as well as Roy et al. (2019), but in contrast to the study recognized by Duong and Lee (2011).

The calculated PLI values for all measured heavy metals were ranged from 1.53 (at site 13) to 3.46 (at site 7) together with the mean value of  $2.24 \pm 0.64$  (Table 5). The findings of the study stated that PLI values were greater than 1.0 over the all studied sites, depicted a major degradation in the study area's terrestrial environmental dominance. The high PLI value may be due to both geo-genic and anthropogenic sources, as well as cumulative contributions of all measured elements (Roy et al. 2019; Rahman et al. 2019). Furthermore, Roy et al. (2019) accomplished a study with higher PLI values in street dust of Delhi, which is evidently similar to the present assessment.

In the current investigation, Cp value for As (0.33) demonstrated unpolluted (lower magnitude of pollution) of street dusts, whereas Cp values of Pb (2.91), Cr (1.96), and Ni (1.82) disclosed moderate level of pollution. Correspondingly, Cp values for Cu (3.37), Zn (4.17), and Cd (179.09) exhibited significantly strong level of pollution indicate severe contamination in street dusts (Fig. S2). Unlike other indices results, Cp revealed the higher level of contamination, which means that Cp poses great threat to biota, and arises from the similar sources of the study areas. Nouri and ABDEM (2016) completed a study at Ait Ammar in Morocco and found highest degree of contamination for Pb and Cu by means of Cp values, which are approximately parallel to the present exploration.

The mean mCD values for sampling sites SD5 (7.90), SD10 (7.93), and SD15 (7.46) represented high extent of pollution; however, SD1 (10.37), SD2 (11.01), SD3 (13.43), SD4 (8.72), SD11 (8.25), SD12 (9.85), and SD14 (9.89) showed evidently very high extent of pollution; in the meantime, SD6 (25.40), SD7 (26.10), SD8 (27.50), and SD9 (17.80) were found to be extremely higher extents of pollution owing to the metal(oid)s in street dusts (Fig. S3). Results of the study stated that for higher extent of mCD values along with spatial variations numerous anthropogenic causes in conjunction with the lithogenic factors are mainly responsible (Fahmida

and Islam 2018). The mCD values varied from 0.35 to 6.80, with an average of 2.35, suggesting that the sites examined were contaminated to a moderate degree (Nouri and ABDEM 2016).

Figure 4 shows the individual toxic unit (TU) of each metal and the number of all toxic units (  $TU_S$  ) in each sampling station for the toxic elements examined in Gazipur’s urban street dust. In the studied sites, Pb, Cu, Cr, Cd, Zn, Ni, and As had average toxic unit (TU) values of 0.448, 0.228, 0.926, 2.584, 0.759, 0.930, and 0.122, respectively (Fig. 4). The study found that TU of toxic metal(oid)s in the street dusts analyzed decreased in the following order: Cd > Ni > Cr > Zn > Pb > Cu > As. The sum of toxic units (  $TU_S$  ) for all sampling sites (except SD13: nearly 4.0) was significantly higher than 4, suggesting moderate to severe toxic metal toxicity to dust-dwelling organisms throughout the study region (Wang et al. 2016; Kormoker et al. 2019; Proshad et al. 2019). For the considered toxic metal(oid)s, the sum of toxic units (  $TU_S$  ) for sampling sites SD6, SD7, SD8, and SD9 were comparatively higher than for other sites, indicating more or less analogous tendencies of metal concentrations in street dusts (Fig. 4 and Table 2).

The Nemerow integrated pollution index (NIPI) of potentially toxic elements in street dusts was evaluated in this report. The NIPIs of all sampling sites were varied from 31.94 to 128.12 with an average of 59.38, indicating that the total emission levels differed greatly between different sites (Fig. S4). Generally, these findings also suggest that the street dusts were polluted by anthropogenic emission (Nouri and ABDEM 2016). Moreover, results of the study revealed that the urban street dust was heavily polluted with higher NIPI values ( $\geq 3$ ). Dust sampling sites with higher pollution levels were predominantly situated in areas heavily influenced by manufactories and heavy congested traffic (Yang et al. 2011). However, these trends may be attributed to urbanization, as well as the distribution of industrial areas, vehicles stoppage and workshops, and commercial areas (Shifaw 2018). Higher level of NIPI values also observed previously,

reported by numerous scientist, for example, Nouri and ABDEM (2016), Yang et al. (2011), and Shifaw (2018).

For the advanced understanding of street dusts contamination and associated ecological risk, both the individual risk factors ( $E_r^i$ ) and cumulative potential ecological risk (PER) are predominantly essential (Kabir et al. 2020b). The mean  $E_r^i$  values of toxic metals decreases following the order: Cd > Cu > Pb > Ni > Zn > Cr > As. The  $E_r^i$  values of Pb, Cu, Cr, Cd, Zn, Ni, and As were in the ranges of 5.3972–14.5327, 6.1449–16.8458, 1.7664–3.9296, 1336.3636–5372.7273, 2.7305–4.1744, 2.5467–7.2941, and 0.9919–3.3363, respectively (Table 6). In view of the influence of single toxic metals on the toxicity state, Cd was the key contributor for ecological risk in the street dusts. However, risk attributable to Pb, Cu, Cr, Zn, Ni, and As were low in all sites of urban areas of Gazipur. Overall, the potential ecological risk of Cd was much higher than those of all measured metals, and Cd should be concerned in terms of risk for terrestrial ecosystem (Aminiyan et al. 2018). Besides, the manufacturing factory is to be expected an important point source for Cd (Pan et al. 2016). Furthermore, phosphatic fertilizers and pesticides factories and shops are found to be weightier sources of some toxic metals as well as for Cd (Sun et al. 2013). Considerably, higher Cd level also found triggered by anthropogenic activities by numerous studies (Aminiyan et al. 2018; Kamani et al. 2017; Sun et al. 2013; Pan et al. 2017).

Moreover, the mean PER value of all sampling sites was decreased from 5418 to 1360, with the mean value of 2520, demonstrating that the comprehensive ecological hazard of the examined toxic metals belonged to the very high hazard level. However, the study’s findings revealed that all sampling sites across the study area posed a potential significant risk to the native environment (Table 6). This study’s findings are consistent with previous assessments of toxic metals in Delhi’s street dust (Suryawanshi et al. 2016), Cairo’s (Abdel-Latif and Saleh 2012), Beijing’s (Tong et al. 2020), and Rafsanjan’s (Aminiyan et al. 2018).

### Human health risk evaluation

In this study, health risks from three exposure pathways to toxic metal(oid)s in street dust were characterized into two extensive portions: non-carcinogenic (NCR) and carcinogenic (CR) health risks. A distinct appraisal was completed for children and adults in every site. This work revealed that ingestion of dust particles appears to be the foremost route of exposure pathway for metal(oid)s to adults and children, followed by dermal contact and inhalation considering non-carcinogenic risk. This finding is a good covenant with the results of several health risk studies (Rahman et al. 2019; Idris et al. 2018; Zheng et al. 2010a). For all the studied toxic metal(oid)s at all sampling sites, the estimated CDI values were found within

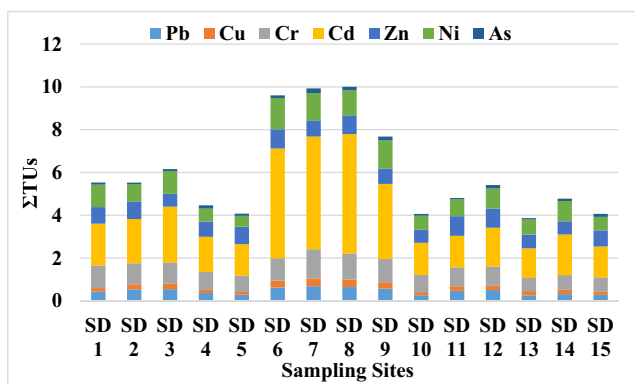


Fig. 4 Toxic unit values of toxic metals in street dusts of urban areas of Gazipur City, Bangladesh



**Table 6** Potential ecological risk factors ( $E_r^i$ ) and potential ecological risk indexes (PER) of toxic metals in street dust

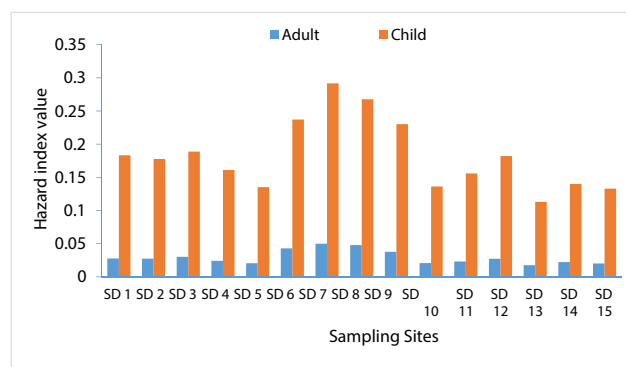
Sites	Potential ecological risk factor ( $E_r^i$ )							Potential ecological risk (PER)	Pollution degree
	Pb	Cu	Cr	Cd	Zn	Ni	As		
SD 1	9.1589	7.5701	3.0464	1881.8182	3.5144	5.2318	1.5329	1911.8727	Very high
SD 2	11.3785	11.0280	2.8256	1990.9091	3.6542	4.1246	1.0821	2025.0020	Very high
SD 3	11.5421	12.3364	2.8384	2509.0909	2.7305	5.3149	1.3526	2545.2058	Very high
SD 4	7.7570	6.7523	2.4224	1581.8182	3.1888	3.0588	2.3445	1607.3420	Very high
SD 5	5.9813	6.1449	2.1792	1418.1818	3.7147	2.5467	1.5329	1440.2815	Very high
SD 6	13.1542	15.2336	3.0048	4936.3636	4.0793	7.2941	1.8936	4981.0233	Very high
SD 7	14.5327	16.8458	3.9296	5072.7273	3.4323	6.3529	3.3363	5121.1569	Very high
SD 8	13.7150	16.2150	3.5104	5372.7273	3.9755	5.8270	2.5248	5418.4949	Very high
SD 9	12.1262	13.6916	3.1296	3381.8182	3.2651	6.5329	2.7953	3423.3589	Very high
SD 10	5.5607	6.6355	2.3200	1445.4545	2.7738	3.2664	1.2624	1467.2734	Very high
SD 11	9.9065	9.9065	2.5472	1418.1818	4.1744	3.9170	0.9919	1449.6253	Very high
SD 12	10.7243	9.0187	2.6048	1745.4545	4.0706	4.6782	2.4346	1778.9858	Very high
SD 13	5.3972	9.2056	1.7664	1336.3636	2.9020	3.5156	0.9919	1360.1423	Very high
SD 14	6.6589	9.1822	1.9872	1827.2727	2.7925	4.7336	1.7133	1854.3404	Very high
SD 15	5.8879	7.5701	1.8848	1390.9091	3.4193	3.1419	2.1641	1414.9771	Very high
Min	5.3972	6.1449	1.7664	1336.3636	2.7305	2.5467	0.9919	1360.1423	Very high
Max	14.5327	16.8458	3.9296	5372.7273	4.1744	7.2941	3.3363	5418.4949	Very high
Avg.	9.5654	10.4891	2.6665	2487.2727	3.4458	4.6358	1.8635	2519.9388	Very high
SD	3.18084	3.58814	0.60671	1466.9149	0.50062	1.4355	0.7130	1475.19204	Very high
Median	9.9065	9.2056	2.6048	1827.2727	3.4323	4.6782	1.7133	1854.3404	Very high

the acceptable range, which indicates that lower possibility to cause public health effect concerns for both adults and children via all exposure pathways (Table S7).

The possible NCR toxic effects posed by toxic metals are generally characterized by calculating HQ (Proshad et al. 2019). The calculated value of HQ of toxic metals in street dust was lower than 1 for all sites and hence it could be demonstrated that the studied metals in street dust had negligible non-carcinogenic risks to adults and children (Table S8). This finding is a good covenant with the results of Idris et al. (2018) and Qadeera et al. (2020) who studied the toxic elements in street dust and also found lower non-carcinogenic risk. In contrast, Rahman et al. (2019) perceived higher HQ values of toxic elements in street dust of Dhaka city, Bangladesh. The total HI value for adults and children for all the studied metals in all sites were also found within the safe level recommended by U.S Environmental Protection Agency (1989), indicating lower possibility of non-carcinogenic risk to adults and children from exposure to street dust (Fig. 5). This study also revealed that children have 4–6 times higher HI values than adults and conforming high vulnerability of the children to risks owing to toxic elements in street dust which is similar to the reported results in literature at other locations (Qadeera et al. 2020; Chen et al. 2017; Zheng et al. 2010b). Children are more vulnerable to the risk of metal exposure because of their

lower body weight, lower tolerance level of pollutants, outdoor play activities, and direct hand-to-mouth exposure of dust (Wahab et al. 2020; Zhao et al. 2014; Luo et al. 2012).

The CR of Ni, As, and Pb via three exposure pathways are presented in Table 7. According to U.S Environmental Protection Agency (1989), any carcinogenic risk value within the range of  $1.00 \times 10^{-6}$  to  $1.00 \times 10^{-4}$  is acceptable. In this study, the CR value was perceived in a decreasing order following the exposure pathway of ingestion > dermal contact > inhalation; this was similar to non-carcinogenic exposure ways in all the sites. This finding is in tune with the results

**Fig. 5** Hazard index (HI) of toxic metals in urban street dusts of Gazipur City, Bangladesh

**Table 7** Carcinogenic risk of adult and children due to ingestion, dermal contact, and inhalation of nickel, arsenic, and lead in street dust

Sites	Ni (Nickel)			Arsenic (As)			Lead (Pb)		
	Ingestion	Dermal contact	Inhalation	Ingestion	Dermal contact	Inhalation	Ingestion	Dermal contact	Inhalation
SD 1	4.66E-09	1.86E-11	1.53E-11	3.74E-07	1.84E-08	1.97E-11	4.89E-08	1.95E-10	1.27E-12
SD 2	3.67E-09	1.47E-11	1.21E-11	2.64E-07	1.30E-08	1.39E-11	6.08E-08	2.42E-10	1.58E-12
SD 3	4.73E-09	1.89E-11	1.55E-11	3.30E-07	1.62E-08	1.74E-11	6.16E-08	2.46E-10	1.60E-12
SD 4	2.72E-09	1.09E-11	8.94E-12	5.72E-07	2.81E-08	3.02E-11	4.14E-08	1.65E-10	1.07E-12
SD 5	2.27E-09	9.05E-12	7.45E-12	3.74E-07	1.84E-08	1.97E-11	3.19E-08	1.27E-10	8.29E-13
SD 6	6.50E-09	2.59E-11	2.13E-11	4.62E-07	2.27E-08	2.44E-11	7.02E-08	2.80E-10	1.82E-12
SD 7	5.66E-09	2.26E-11	1.86E-11	8.15E-07	4.00E-08	4.29E-11	7.76E-08	3.10E-10	2.01E-12
SD 8	5.19E-09	2.07E-11	1.70E-11	6.16E-07	3.03E-08	3.25E-11	7.32E-08	2.92E-10	1.90E-12
SD 9	5.82E-09	2.32E-11	1.91E-11	6.82E-07	3.35E-08	3.60E-11	6.47E-08	2.58E-10	1.68E-12
SD 10	2.91E-09	1.16E-11	9.55E-12	3.08E-07	1.51E-08	1.62E-11	2.97E-08	1.18E-10	7.71E-13
SD 11	3.49E-09	1.39E-11	1.15E-11	2.42E-07	1.19E-08	1.28E-11	5.29E-08	2.11E-10	1.37E-12
SD 12	4.17E-09	1.66E-11	1.37E-11	5.94E-07	2.92E-08	3.13E-11	5.73E-08	2.28E-10	1.49E-12
SD 13	3.13E-09	1.25E-11	1.03E-11	2.42E-07	1.19E-08	1.28E-11	2.88E-08	1.15E-10	7.48E-13
SD 14	4.22E-09	1.68E-11	1.38E-11	4.18E-07	2.05E-08	2.20E-11	3.56E-08	1.42E-10	9.23E-13
SD 15	2.80E-09	1.12E-11	9.19E-12	5.28E-07	2.59E-08	2.78E-11	3.14E-08	1.25E-10	8.16E-13

of Qadeera et al. (2020) who assessed health risk assessment of toxic elements in street dust from two urbanized cities of Pakistan. The calculated carcinogenic risks for adults and children at all sites via ingestion, dermal contact, and inhalation were within the acceptable range; therefore, it may demonstrate that there would not happen serious long-term health hazards for adults and children with respect to Ni, As, and Pb in street dust samples. Similar result was also found by Adamiec and Jarosz-Krzemińska (2019) and Taşpınar and Bozkurt (2018) who studied the heavy metals in street dust and also found inferior carcinogenic risk. However, possibility of occurring non-carcinogenic and carcinogenic health risks is low but long-term exposure of toxic metals from street dust in the study area cannot be overlooked.

### Conclusions

A set of high-quality data on potentially toxic elemental abundances in street dust samples from an urban city (Gazipur, Bangladesh) of a developing country has been comprehensively evaluated. This city has undergone rapid industrialization, unplanned urbanization, and loaded with seriously high traffic. In terms of the systematic sampling and concomitantly analyzed elemental abundances, pollution scenario, elemental distributions, source apportionment, ecological, and probabilistic health risks conditions are briefly summarized as follows:

1. Potentially toxic elemental abundances of Pb, Cu, Cd, Cr, Zn, Ni, and As were relatively higher than the

recommended limits where Cd and As obtained the highest and lowest enrichments, respectively. However, mean abundances of measured metal(oid)s followed the descending order as Zn > Cr > Cu > Pb > Ni > Cd > As.

2. Descriptive statistics (coefficient of variation: 14.5–59.0%) clearly indicated an inhomogeneous distribution of the measured elements. Concomitantly, inverse distance weighting (IDW) maps displayed that enrichment of measured elements are relatively higher in the sampling sites with high loadings of traffic, population, industrialization, and urbanization.
3. Multivariate statistical approaches (Pearson’s correlation coefficient analysis, principal component analysis, and cluster analysis) identified several point and non-point sources for the studied potentially toxic elements. Transportation-affiliated activities and industrial processes were mostly governing the elemental enrichments rather than the natural processes.
4. Element-specific environmental indices (geo-accumulation index, enrichment factor, and contamination factor) followed the descending order as Zn > Cr > Cu > Pb > Ni > Cd > As, whereas depending on the toxicity response factor individual ecological risks followed the descending order as Cd > Cu > Pb > Ni > Zn > Cr > As. However, sampling site-specific composite pollution and potential ecological risks are significantly higher in the sampling sites loaded with the transportation affiliated activities and industrial processes. Most significantly, the whole studied site possesses high potential ecological risks.
5. An evaluation of probabilistic non-carcinogenic and carcinogenic health risks through different exposure

pathways (ingestion > dermal contact > inhalation) demonstrated insignificant health risks.

Above findings and affiliated assessment invoke that similar study sites around the globe should be monitored and evaluated on a regular basis to ensure the safe public health and ecological sustainability.

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## Declarations

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