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Spatial distribution, source analysis, and health risk assessment of heavy metals contamination in house dust and surface soil from four major cities of Nepal

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ABSTRACT

Raising population, deteriorating environmental conditions and limiting natural resources to handle the key environmental health problems have critically affected human health and the environment. Policy makers and planners in Nepal are more concerned today than at any other time in the past about the deterioration of the environmental condition. Therefore, understanding the connection between pollution and human wellbeing is fundamental endeavors to control pollution exposures and secure human wellbeing. This ability is especially critical for countries like Nepal where the issues of environmental pollution have customarily taken a second place to request for economic development. In this study, spatial distribution and sources of 12 heavy metals (HMs) were investigated in surface soils (n=24) and house dust (n=24) from four major urban areas of Nepal in order to mark the pollution level. Additionally, a health risk was estimated to establish the link between HMs pollution and human health. Results showed that the median concentration of Ag, Cd, Co, Cr, Cu, Ni, Pb, Sb, Mn and Zn in soil and dust were 2–13 times greater than the background value. The As, Zn, Cu, Cd, and Pb showed a relatively higher spatial variability in soil and dust. Zn was the most abundant metal measured in dust and soil and accounted for 59% and 55% of $\sum_7$HMs, respectively. The HMs in soil and dust were poorly correlated with total organic carbon (TOC) and black carbon (BC), suggesting little or no influence on HMs contamination. Source analysis study indicated the distribution of Cr, Ni, Sb, Ag, Pb, Cu, and Zn in soil and dust are mainly affected by anthropogenic sources, particularly traffic emissions, industrial source, and domestic households materials, while Co, Fe, As, Mn and Cd were from natural sources. The estimated carcinogenic risk (CR) of HMs in soil and dust exceeded the acceptable level of human exposure, recommending significant CR to the local population.

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1. Introduction

Heavy metals (HMs) are characteristic constituents of the Earth's crust. However, human activities have significantly altered the balance and biochemical and geochemical cycles of some essential HMs (Kabata-Pendias and Mukherjee, 2007). Heavy metal pollution of soil is one of the significant environmental problems today due to rapid urbanization and industrialization (Gowd et al., 2010). They could pose a serious threat to humankind and the environment because of their toxicity, persistence behavior and bio-accumulation and bio-magnification characteristics in the food chain (Pandey et al., 2003). Therefore, the environmental risk evaluation of the contaminated soil is of specific concern for agricultural and non-agricultural areas (Mc dowell et al., 1993; Moreno et al., 1994; Grzebisz et al., 2001).

Soil contamination can be local or diffuse, which could be a sink of HMs. Local soil contamination in the urban area occurs through

industrial activities, traffic emission, inadequate waste disposal, fuel combustion and mining activities (Li et al., 2001; Chen et al., 2005; Zhang et al., 2005; Shen et al., 2008; Bullock and Gregory, 2009). Urban soil is a crucial part of urban environments, contributing directly or by implication to the general quality of life for city residents. It is also a key part of the biogeochemical cycle, filtration of water, supporting plants and frameworks, and numerous recreational exercises (Stroganova et al., 1997). Additionally, HMs can be transported to a longer distance through atmospheric particulate matter after their release into the environment (Wu et al., 2010). Likewise, a portion of atmospheric metals can equally be exchanged to soils by atmospheric deposition (Lu et al., 2010). In this way, urban soils are the critical sinks and source for HMs pollution. Heavy metal in urban soils has been appeared to be extremely important indicators of environmental contamination. A number of investigations have detailed worldwide contamination of HM in urban soils (Cannon and Horton, 2009; Dartan et al., 2015; Hossain et al., 2015; Jiang et al., 2017; Devi and Yadav, 2018). The heavy metals, for instance, Pb, Cu, Zn, and Cd are the some of the more commonly studied inorganic contaminants in urban soils, while As received less attention more probably due to analytical limitations, despite its major public health con-

cern. Thus, understanding the contamination levels and regional variation of HMs in various ecological frameworks is vital for pollution control and urban planning.

Household dust is another important ecological framework that is capable of adsorbing different types of pollutants because of their large surface areas. Urban house dust serves as both the sink as well as the source of HM pollution, which can show the attributes of HMs distribution and accumulation in urban condition. The pollutants once adsorbed to dust; they become non-degradable or degrade at a rate moderately slower than those adsorbed on outdoor dust (Ong Ayoko et al., 2007). Household dust may collect adequate measures of contaminants including HMs that could be of the severe health hazard to the potentially sensitive subpopulations such as newborn babies and elderly living in the house (Beamer et al., 2008; Latif et al., 2009). The deposition velocities of the fine or little particles are relatively lower than those of coarse particles, and consequently remain suspended in the air for the longer duration, leading to human respiration problems (Bodin et al., 2000; Matson, 2005). In this manner, household dust is capable of collecting different contaminants including HMs and other comparative contaminants. HM pollution in household dust is of special concern since individuals spend substantial periods of time every day in indoor condition including homes, workplace, and schools (Ong Ayoko et al., 2007; USEPA, 2011). Thus, the analysis of HMs in house dust provides an important information for assessing the chronic exposure of indoor pollution.

Raising population, deteriorating environmental conditions and limiting environmental resources to handle the key environmental health problems have significantly affected human well-being and the health of the environment (Frangi and Richard, 1997; Huang et al., 2007; Solgi et al., 2012; USEPA, 2014). Policy makers, planners and/or the persons related to formulating policies in Nepal are more concerned today than any other time in the past about the deterioration of environmental condition. Therefore, the ability to interface health and environmental data, and thereby to comprehend the connection between the levels of pollution exposure and well-being result, is obviously fundamental in endeavors to control pollution exposures and secure human health. This ability is especially critical for countries like Nepal where the issues of environmental pollution have customarily taken a second place to request for economic development (UNEP/WHO, 1996). This study aims to investigate the spatial distribution and sources of HMs in soil and house dust from four major urban areas (viz. Kathmandu, Pokhara, Birgunj, and Biratnagar) of Nepal. Additionally, human health risk was estimated to predict the dangerous of HMs contamination on the children and adult population.

2. Materials and methods

2.1. Study area

Four major cities of Nepal (i.e. Kathmandu, Pokhara, Birgunj, and Biratnagar) were chosen for the collection of surface soil and house dust samples (Fig. S1). Surface soil (n=24) and house dust samples (n=24) were collected from various ecological compartments during August–October 2014. The details about study sites and sampling locations are discussed previously (Yadav et al., 2017a, 2018a).

2.2. Sample collection

Surface soil and house dust samplings in four Nepalese cities were conducted as discussed in our previous studies (Yadav et al., 2017b, 2018b, c). Briefly, about 50 g of surface soils (0–15 cm depth, vegeta-

tion removed) at 24 sites in the 4 selected urban center of Nepal were collected during August–October 2014 utilizing stainless steel scoops. Each soil sample was a composite of 3 sub-samples which was collected and mixed from a radius of 5 m in a different direction. The soil samples were wrapped in the aluminum foil packed into polythene zipper bags and was transported to the laboratory keeping in the ice bag. Later, the soil samples were freeze-dried, ground to powder, sieved through 500 μm sieves, and stored at −20 °C until chemical analysis.

Likewise, a sum of 24 house dust was collected from the indoor condition, representing residential, occupational, and public environments. Dust samples were collected by sweeping of kitchen room, study room, bedroom, living room, office and passage of the households. Dust samples were collected using vacuum cleaner made for the domestic purpose. A separate paper dust bags were used to collect dust samples from the individual household. About 50 g of dust was collected and packed in polythene zipper bag before transporting to the laboratory. In the laboratory, the dust samples were homogenized, sieved with a mesh size of 500 μm and stored at −20 °C until chemical analysis.

2.3. Heavy metal analysis

Both the soil and dust samples were acid digested in HNO_3-$HClO_4$-HF following standard protocol discussed previously (Pansu and Gautheyrou, 2006). Briefly, 0.5 g of freeze-dried, homogenized and sieved soil/dust was taken in a Teflon vessel and digested with 4 mL of concentrated HNO_3, 1 mL of $HClO_4$, and 2 mL of HF at 140–160 °C and 0.8 MPa. After cooling at ambient temperature, the clear solution was transferred to a 25 mL volumetric flask and makeup with ultrapure distilled water. All the glassware used in the chemical analysis were cleaned and decontaminated in a 5% nitric acid solution for 24 h, followed by rinsing with distilled water. All the acids used in the study were of guaranteed reagents (GR). The digested samples were then filtered through a 0.45 μm filter. The total Cr, Mn, Fe, Co, Ni, Cu, Zn, Ag, Cd, Sb, and Pb content in the soil/dust were analyzed by inductively coupled plasma-atomic emission spectrometer (ICP-AES, Optima 3300DV, Perkin Elmer, USA). Total As in soil/dust was determined by Hydride Generation Atomic Fluorescence Spectrometry (HG-AFS, SA-10, Beijing Titan Instruments Co., Ltd., China).

2.4. Quality assurance and quality control (QA/QC)

Before sample collection and analysis, all the glassware and equipment were cleaned with 10% HNO_3. The analytical precision and accuracy of ICP-AES were accomplished by analyzing 6 sets of standard reference materials (SRM). SRM of soil (GSS-24) were obtained from the Centre of National Standard Reference Material of China and were analyzed in a similar way as the soil/dust sample. Good agreement was achieved between determined value and certified value with recovery rate ranging from 72 to 125%. The metals such as Cd, Co, Cr, Cu, Ni, Pb, Zn, and As showed better recovery rate (94–125%), while Mn, Sb, and Ag had low recovery (72–75%) (Table S1, Supplementary information). A total of 6 blank samples were analyzed following similar analytical process utilized for sample analysis. The concentration of metals in blank were either not detected or detected in the negligible amount. The relative standard deviation was below ±5%. The method detection limit (MDL) was calculated as the 3 times standard deviation plus mean of all the blanks. In the case of non-detection of metal in blanks, the MDL was calculated as the 3 times signal to noise ratio (S/N) of the lowest spiked

standard. The MDLs of metals (mg/kg) in soil/dust samples were Ag (0.23), Cd (0.011), Co (0.052), Cr (5.01), Cu (0.078), Ni (0.092), Pb (0.11), Sb (0.065), Fe (3.31), Mn (5.05) and Zn (0.043) in ICP-AES method; and As (0.01) in HG-AFS method. Metal concentration in soil and dust were expressed on dry weight (dw) basis.

2.5. Enrichment factor (EF)

Enrichment factor (EF) is used to assess the enrichment degree of metal in soil or dust. EF can also help to differentiate an anthropogenic source from and natural sources (Han et al., 2006; Liu et al., 2014). More commonly, EF is also utilized to evaluate the level of human impact. EF is calculated by dividing its ratio to the reference element by a similar proportion found in the background (Buat-Menard and Chesselet, 1979).

$$EF = \frac{\left(C_i/C_{ref}\right)_{sample}}{\left(C_i/C_{ref}\right)_{background}} \qquad (1)$$

where Ci is the concentration of individual elements in soil/dust (mg/kg); C ref is the concentration of reference element for normalization (mg/kg). Generally, Al, Fe, Mn, Ti, and Sc are used as the reference element in the estimation of EF (Kara et al., 2014; Amil et al., 2016; Hsu et al., 2016). In this study, we used Mn as a reference element due to its relatively higher concentration and more stability in the earth crust (Tasdemir and Kural, 2005; Han et al., 2006; Cheng et al., 2018). The background value of HMs was adopted from Tasdemir and Kural (2005) because the background concentration of HMs is not available in the case of Nepal. The EF value near to unity indicates natural origin, those less than unity suggests possible mobilization/depletion of metal (Zsefer et al., 1996; Bhuiyan et al., 2010), while EF value greater than unity recommends that the element is of anthropogenic origin. Likewise, EF greater than 10 is indicative of non-crusted source.

2.6. Statistical analysis

Statistical analysis (min, max, median) of the HMs data were analyzed using IBM SPSS (ver 21) software. Spatial distribution map of HMs was made using ESRI-Arc GIS geospatial (ver 10.3) software. Pearson's correlation and principal component analysis (PCA) were applied to investigate the sources of HMs in the soil. Factor analysis (FA, the components of the PCA) was performed by Varimax rotation. Varimax rotation was used in light of the fact that orthogonal rotation minimizes the number of factors with a high loading on each component and hence encourages the elucidation of PCA results.

2.7. Health risk assessment

Rapid urbanization is a universal phenomenon and thus turns out to be remarkably critical to human well-being and prosperity (Davydova, 2005). HM contaminants in soils and house dust have been reported to cause serious health impact to human and the environment (Zheng et al., 2010; Mohmand et al., 2015; Teng et al., 2015). The surface soil and house dust which are (usually 63 µm in size) easily suspended into the atmosphere and have the tendency to absorb by the human through ingestion, inhalation, and dermal contact. Consequently, it is important to assess the degree of dangers to potential receptors. Because of proximity to large population, the contaminated urban soils and house dust can directly pose a huge health risk through soil ingestion, dust inhalation, or dermal contact

(Abrahams, 2002; Siciliano et al., 2009). In this study, the carcinogenic and non-carcinogenic risk of HMs in soil/dust was estimated according to the United States Environmental Protection Agency (USEPA) health risk assessment model (USEPA, 2013). Inhalation, ingestion, and dermal contact are the three critical routes of HMs exposure to the human. The average daily dose (ADDs, mg/kg/day) of the HMs via soil/dust inhalation, ingestion, and dermal contact, as exposure pathways can be estimated by following Eqs. (2)–(4).

$$ADD_{ingest} = \frac{C \times IgR \times EF \times ED \times CF}{BW \times AT} \qquad (2)$$

where C is the concentration of the HMs in soil (mg/kg) or dust (mg/kg); IgR is the ingestion rate of soil/dust (mg/day); EF is the exposure frequency (320 days/yr); ED is the exposure duration; CF is the conversion factor (1×10^{-6} kg/mg); BW is body weight (80 kg for adult and 18.6 kg for children); and AT is the average lifetime.

$$ADD_{dermal} = \frac{C \times SA \times AF \times ABS \times EF \times ED \times CF}{BW \times AT} \qquad (3)$$

where SA is the exposed skin area (cm²); AF is the skin adherence factor for soil/dust (mg/cm), and ABS is the dermal absorption factor from the soil/dust (chemical-specific);

$$ADD_{inhale} = \frac{C \times IhR \times EF \times ED \times \frac{1}{PEF}}{BW \times AT} \qquad (4)$$

where IhR is the inhalation rate (m³/day); and PEF is particulate emission factor (1.36×10^9 m³/kg).

The hazard quotient (HQ) is employed to estimate the non-carcinogenic risks of HMs in soils/dust in different exposures. It is the ratio of the ADD and the specific reference dose (RfD) and can be estimated using the following equation (5).

$$HQ = \frac{ADD}{RfD} \qquad (5)$$

RfD (mg/kg/day) is the daily maximum allowable dose of HMs without posing the non-carcinogenic risk to human during life-span. Three different types of RfDs are used for three different exposure pathways: reference dose (RfD$_o$, (mg/kg/day) for ingestion, RfD$_{ABS}$, (mg/kg/day) for dermal contact and RfD$_i$ (mg/m³) for inhalation.

The total risk of specific chemicals through multiple exposures is expressed as the hazard index (HI). The total risks of HMs in soil/dust through multiple exposures can be calculated as per equation (6).

$$HI = \sum HQ_i \qquad (6)$$

where i = different exposure pathways.

The carcinogenic risks to humans are estimated using the average daily doses (ADD) multiplied by respective slope factor (SF). A slope factor is an upper bound probability of an individual developing cancer as a result of a lifetime exposure to an agent by ingestion or inhalation.

$$Cancer\ risk\ (CR) = ADD \times SF \qquad (7)$$

CR value less than 1×10^{-6} specifies negligible carcinogenic risk, while CR greater than 1×10^{-4} recommends high carcinogenic risk to

human (Wu et al., 2015). The specific and constant parameters used in health risk assessment model are given in Table S2, Supplementary Information.

3. Results and discussion

3.1. Overall comments on metal concentration

The statistical characteristics of major metals together with seven priority HMs (As, Cd, Cr, Cu, Ni, Pb, and Zn) measured in surface soil and house dust in this study are summarized in Table 1. The median concentration of Ag, Cd, Co, Cr, Cu, Ni, Pb, Sb, Mn and Zn in the soil were 2–10 times greater than the background concentration in shale (Turekian and Wedepohl, 1961), suggesting either contamination or influence of pedogenic factors (Ma et al., 1997). Likewise, these metals in dust were 2–13 times greater than the background concentration (Turekian and Wedepohl, 1961). Only As and Fe showed lower concentration than the background concentration. The seven priority HMs measured in dust were slightly higher than those in the soil, and ranged from 456 to 5620 mg/kg dw (median 2120 mg/kg dw) and 982–6140 mg/kg dw (median 1870 mg/kg dw), respectively. Zn was the most abundant metal measured both in dust and soil and accounted for 59% and 55% of $\sum_7$HMs, respectively. The abundance of Zn in soil has been previously reported from Bhaktapur, Nepal (Kayastha, 2014). The median concentration of seven priority HMs in dust decreased in the order of Zn>Cu>Pb>Cr>Ni>As>Cd, while they were arranged in decreasing order Zn>Cr>Cu>Pb>Ni>As>Cd in soil. This decreasing trend of metal is consistent with the previous study from four counties of south China (Lia and Jia, 2018).

3.2. Metal contamination in surface soil

Results of the major metals together with seven priority HMs measured in the soil in four urban centers of Nepal are summarized in Table S3 and Fig. 1. Highest concentrations of $\sum_{12}$HMs were measured in Biratnagar and Birgunj with the lowest concentration in Kathmandu and Pokhara. The concentration of $\sum_{12}$HMs ranged from 4470 to 6160 mg/kg dw (median 5060 mg/kg dw), 4180–8000 mg/kg dw (median 5210 mg/kg dw), 2860–5770 mg/kg dw (median 4230 mg/kg dw) and 2370–9950 mg/kg dw (median 4700 mg/kg dw) in Biratnagar, Birgunj, Kathmandu and Pokhara, respectively. The concentration $\sum_7$HMs were slightly greater in Kathmandu and

Pokhara than those in Birgunj and Biratnagar, and ranged from 1440 to 2500 mg/kg (median 1980 mg/kg dw), 1040–6130 mg/kg dw (median 2000 mg/kg dw), 1230–3180 mg/kg dw (median 1600 mg/kg dw), and 1450–2360 mg/kg dw (median 1750 mg/kg dw), respectively. Although the concentration of HMs showed some variation among different Nepalese cities, they were statistically insignificant ($p < 0.05$). The concentrations of $\sum_{12}$HMs measured in the soil in this study were compared with similar studies reported worldwide and are presented in Table S4. The average median concentration of Cu, Cr, Pb, Zn and Ni in soil exceeded the respective concentration reported worldwide, while the concentrations of Cd and As measured in this study were consistent with previous studies (Odewande and Abimbola, 2008; Jiang et al., 2017) (Table S4). The median concentrations of Zn measured in soil in this study were 10–50 times greater than global studies (Chen et al., 2005; Hossain et al., 2015; Vieira da Silva et al., 2016; Jiang et al., 2017; Mazurek et al., 2017), but comparable to study from Islamabad in Pakistan (1640 mg/kg) (Ali and Malik, 2011). The concentration of Cr (279–331 mg/kg dw) and Ni (100–130 mg/kg dw) in the soil in this study were consistent with a study from Turkey (97 mg/kg and 301 mg/kg, respectively) (Dartan et al., 2015). This enrichment of HMs in surface soil could be because of rapid urbanization and poor management of industrial effluents (Lia and Jia, 2018). In summary, the comparison of the HMs concentration in this study with global studies demonstrated that the concentration of seven priority HMs were several folds higher than those reported worldwide with exception of As and Cd. Similar pattern of priority HMs in soil has been reported previously from Chinese cities (Jiang et al., 2017; Lia and Jia, 2018).

The profile of the metals including seven priority HMs measured soil in this study is shown in Fig. 2 and S2. In general, Mn was the most abundant metal measured in soil, followed by Fe, Zn, and Cr, and accounted for 27–45%, 21–26%, 17–25%, and 5–7% of the $\sum_{12}$HMs, respectively. Among seven priority HMs, Zn was identified as the most abundant metal irrespective of the study area, followed by Cr, Cu, and Pb, and accounted for 51–55%, 16–19%, 15–16% and 7–8% of $\sum_7$HM, respectively. As and Cd were lowly detected in soil and accounted for less than 1% of $\sum_7$HMs, respectively.

The spatial distribution of five priority HMs measured in the soil in this study is shown in Fig. 3 and S3. The concentrations of As and Cd in soil were particularly low to be plotted on arc GIS spatial map. Hence, we dropped these two metals, and only five priority HMs was used for plotting spatial distribution map. It is clear from Fig. S3 that the highest concentration of $\sum_7$HMs was observed at PKS-3 (5280 mg/kg dw), PKS-5 (4600 mg/kg dw) and PKS-1 (2160 mg/kg

Table 1
Statistical characteristic of heavy metals in house dust (mg/kg) and surface soil (mg/kg) of Nepal.

| mg/kg	House dust				Surface soil				[a]Background conc.
	Min	Max	Median	SD	Min	Max	Median	SD	
Ag	12.6	141	36.5	26.0	27.2	93.0	60.9	15.9	13
Cd	0.5	55.8	1.8	11.0	0.4	2.7	1.1	0.6	0.3
Co	5.6	46.1	28.1	10.5	19.5	59.4	46.0	9.3	19
Cr	76.2	712	231	134	135	393	309	55.2	90
Cu	28.5	1250	275	293	132	1010	277	163	45
Ni	22.2	210	122	54.3	47.2	154	121	24.2	68
Pb	15.6	1070	233	207	59.8	294	137	59.8	20
Sb	1.5	17.5	5.8	3.7	1.7	10.0	6.9	1.7	1.5
Fe	319	1070	838	160	370	1570	1190	275	47,200
Mn	548	3480	1650	613	606	3360	1870	683	850
Zn	313	2300	1260	570	606	4260	1020	812	95
As	0.5	13.8	3.0	2.6	1.5	27.7	4.5	5.7	13
$\sum_{12}$HM	1340	10,400	4680	2080	2000	11,200	5040	2100	–
$\sum_7$HM	456	5620	2120	1270	982	6140	1870	1120	–

[a] Turekian and Wedepohl (1961).
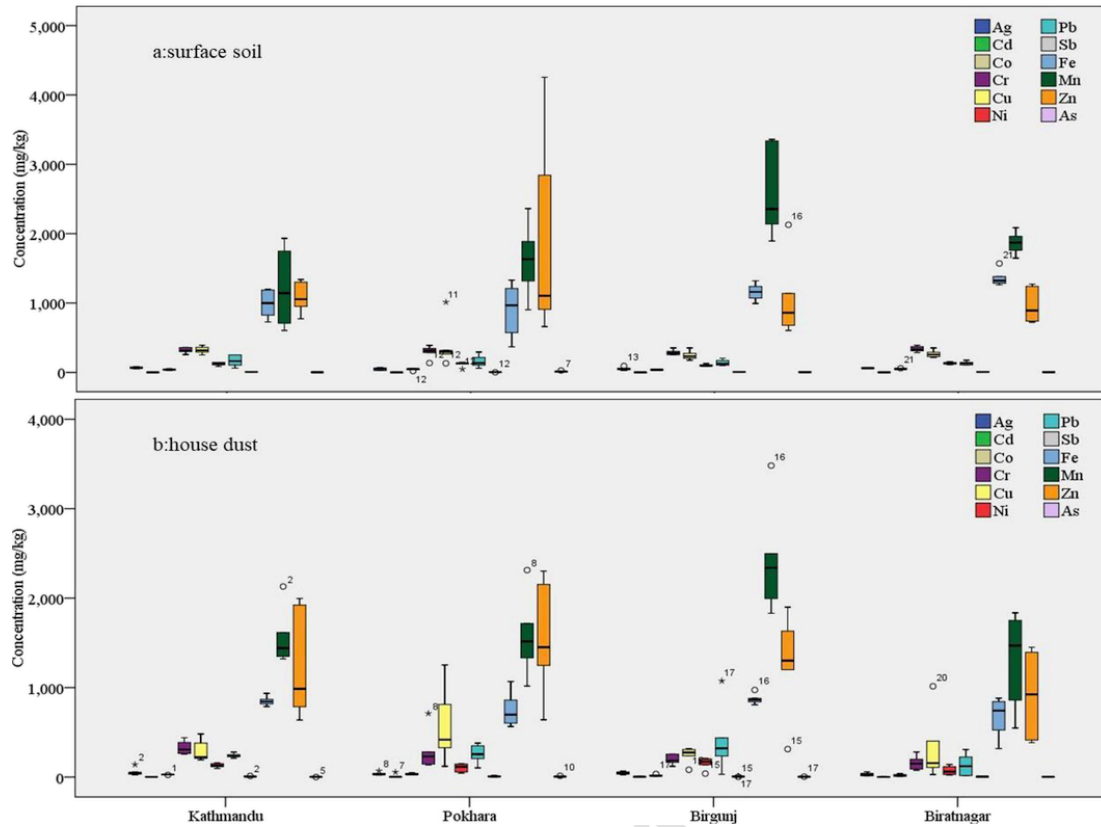
Fig. 1. Box and whisker plots showing the concentration of individual metals in (a) surface soil and (b) house dust from Nepal. The central box represents the concentration from 25 to 75 percentile. The middle bold line represents the median value. The asterisk/stars are the extreme outliers that are >3-times beyond interquartile range.

dw) in Pokhara, BRS-4 (3170 mg/kg dw) in Birgunj, respectively. Likewise, higher concentrations of $\sum_7$HMs were more prevalent at BRT-1 (2190 mg/kg dw) and BRT-5 (2290 mg/kg dw) in Biratnagar, and KTS-1 (2340 mg/kg dw), KTS-3 (2080 mg/kg dw), and KTS-4 (2360 mg/kg dw) in Kathmandu, respectively. PKS-3, BRS-4, and KTS-1 are close proximity to high traffic area; PKS-5 and KTS-3 are the industrial areas, while PKS-1 is an important tourist area. The lowest level of $\sum_7$HMs was measured at PKS-6 (1050 mg/kg dw) in Pokhara, and BRS-5 (1250 mg/kg dw) in Birgunj, which are commercial and residential areas, respectively. Individually, Zn was the most identified metal measured at all sites, irrespective of the study area. The concentration of seven priority HMs followed following the decreasing order in soil:

Cd: Kathmandu>Pokhara>Birgunj>Biratnagar;
Cr: Biratnagar>Kathmandu>Pokhara>Birgunj;
Cu: Kathmandu>Pokhara>Biratnagar>Birgunj;
Ni: Biratnagar>Pokhara>Kathmandu>Birgunj;
Pb: Kathmandu>Biratnagar>Pokhara>Birgunj;
Zn: Pokhara>Kathmandu>Biratnagar>Birgunj; and
As: Pokhara>Birgunj>Biratnagar>Kathmandu;

The coefficient of variation (CV) of the metals was used to compare the variability of metals with respect to space. In this study, the CV of the metals ranged from 17.8 to 88.4%. Among metals, five metals (As, Zn, Cu, Cd, and Pb) showed relatively higher spatial variability in the soil. This implies that both area-source and point source pollution may take place simultaneously. Arsenic showed highest CV (88.4%) among all metals, suggesting the greatest variation in soil samples. The Cr had the lowest CV with a score of 17.8%, indicating weak variation and constant distribution across the study area. This

finding is consistent with the previous study from Southeast China (Cai et al., 2015).

3.3. Metal contamination in household dust

The statistical summary of individual metal together with seven priority HMs analyzed in dust in this study are summarized in Table S5. Unlike soil, Birgunj and Pokhara showed the highest level of $\sum_{12}$HMs in the dust, while Kathmandu and Biratnagar had the lowest concentration. The concentration of $\sum_{12}$HMs ranged from 3270 to 8340 mg/kg dw (median 5530 mg/kg dw), 2680–8380 mg/kg dw (median 4770 mg/kg dw), 3560–6620 mg/kg dw (median 4250 mg/kg dw), and 1420–6030 mg/kg dw (median 3670 mg/kg dw) in Birgunj, Pokhara, Kathmandu and Biratnagar, respectively. Likewise, the concentration of $\sum_7$HMs was relatively higher in Pokhara and Birgunj than those in Kathmandu and Biratnagar, and ranged from 1050 to 4870 mg/kg dw (median 2480 mg/kg dw), 591–3770 mg/kg dw (median 2270 mg/kg dw), 1390–3360 mg/kg dw (median 1880 mg/kg dw), and 530–3210 mg/kg dw (median 1420 mg/kg dw) in Pokhara, Birgunj, Kathmandu, and Biratnagar, respectively (Fig. 1). Birgunj is the gateway of Nepal as most of its trade with India through an open border, while Pokhara is the second largest cities and one of the most enjoyable tourist places in the world. The concentrations of metals measured in house dust in this study were compared with similar studies around the globe and are summarized in Table S6. The median concentration of Cr, Ni, and Pb measured in dust in this study was 1–2 times greater than those reported worldwide (Chattopadhyay et al., 2003; Turner and Simmonds, 2006; Hassan, 2012; Yoshinaga et al., 2014; Cheng et al., 2018), but were comparable to Rasmussen et al. (2013) and Huang et al. (2014). The concentration of Pb mea-
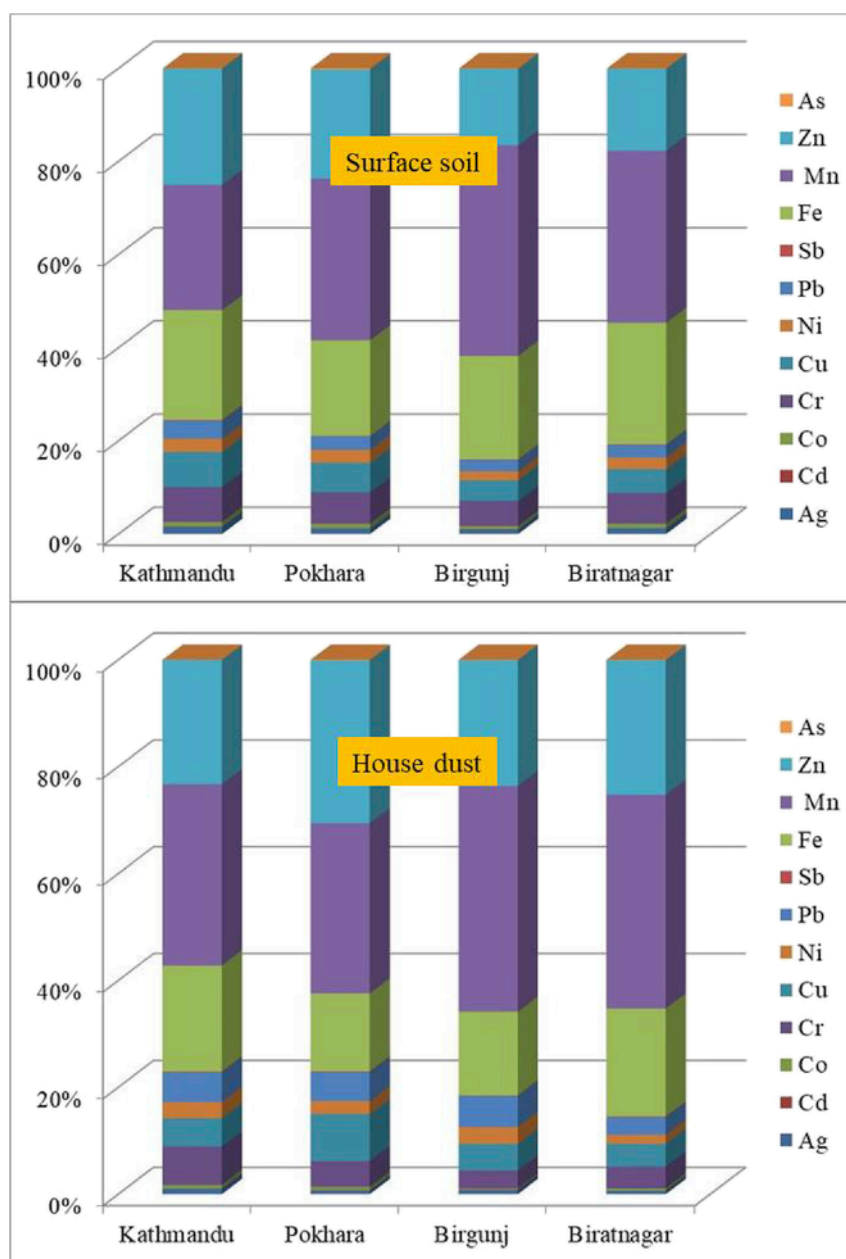
Fig. 2. Profile of heavy metals showing average median % contribution of individual metal to total metal.

sured in house dust in this study was 2–3 folds lower than those reported in Guangzhou in China (Huang et al., 2014) and Selangor in Malaysia (Latif et al., 2009). In summary, the household dust in Nepalese cities contains the same proportion of HMs as with larger megacities in the world, despite thinly populated and low environmental impact in term of urbanization.

The profile of metals measured in house dust from four major urban areas of Nepal is shown in Fig. 2 and S2. The metal profiling depicted in Fig. 2 indicated that Mn was the most abundant metal in the dust as it was in the case of soil, and accounted for 32–42% of $\sum_{12}$HMs. Zn was the second most abundant metal measured in the dust, followed by Fe and Cu, and accounted for 23–30%, 15–20% and 4–9% of $\sum_{12}$HMs, respectively. Among seven priority HMs, Zn was identified as the most abundant metal in the dust, followed by Cu, Cr, Pb, and Ni, and accounted for 52–65%, 11–17%, 8–16%, 9–

14% and 4–8% of $\sum_7$HMs, respectively. A similar pattern of HMs in house dust has been reported from Chengdu in China (Qiao et al., 2013; Chen et al., 2016; Cheng et al., 2018).

The spatial distribution maps of five priority HMs measured in house dust in this study are shown in Fig. 4. Only five out of seven priority metals were used to draw the spatial map as the concentrations of As and Cd were too low. The spatial distribution of rest metals is shown in Fig. S3. The highest concentration of $\sum_7$HMs was observed at PKD-1 (3740 mg/kg dw) and PKD-5 (3550 mg/kg dw) in Pokhara, and BRD-5 (3700 mg/kg dw) in Birgunj, respectively. Moreover, the higher concentrations of $\sum_7$HMs were measured at KTD-2 (3240 mg/kg dw) and KTD-5 (3030 mg/kg dw) in Kathmandu, PKD-2 (2820 mg/kg dw) and PKD-3 (3320 mg/kg dw) in Pokhara, BRD-4 (2670 mg/kg dw) in Birgunj, and BID-2 (2990 mg/kg dw) in Biratnagar, respectively. PKD-1 and PKD-5 are an impor-
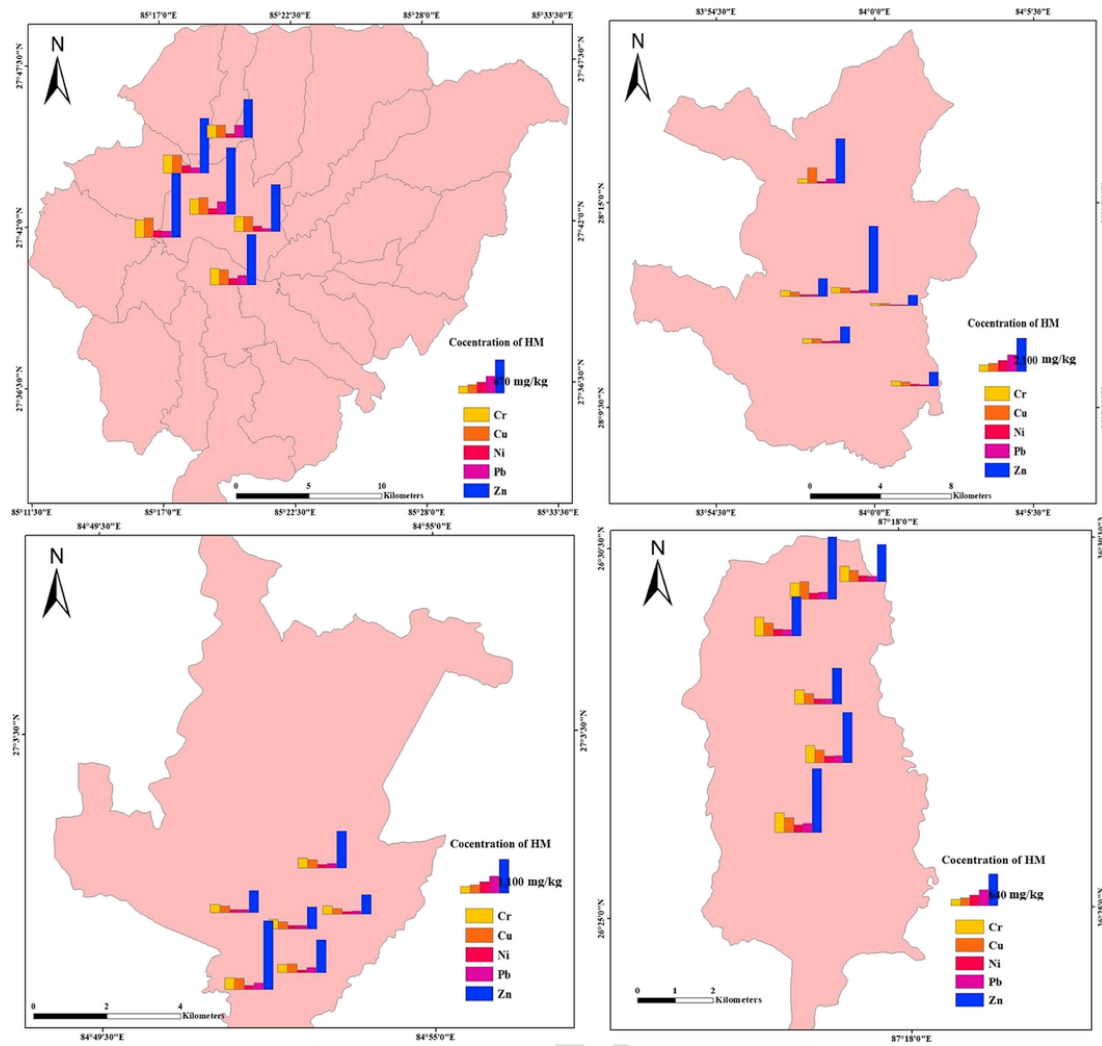
Fig. 3. Spatial maps showing the distribution of priority heavy metals in surface soil (mg/kg) from four major cities of Nepal.

tant tourist area and urban residential area, respectively, while PKD-3 is high traffic area. Likewise, KTD-2 and PKD-2 are close proximity to airport; KTD-5, BRD-5, and BID-2 are a residential area. The lowest concentrations of $\sum_7$HMs were measured at BRD-3 (591 mg/kg dw) in Birgunj and BID-4 (602 mg/kg dw) in Biratnagar which are residential and commercial areas, respectively. The concentration of $\sum_7$HMs in dust followed the following decreasing order:

Cd: Birgunj > Pokhara > Kathmandu > Biratnagar;
Cr: Kathmandu > Pokhara > Birgunj > Biratnagar;
Cu: Pokhara > Birgunj > Kathmandu > Biratnagar;
Ni: Birgunj > Kathmandu > Pokhara > Biratnagar;
Pb: Birgunj > Pokhara > Kathmandu > Biratnagar;
Zn: Pokhara > Birgunj > Kathmandu > Biratnagar; and
As: Pokhara > Birgunj > Biratnagar > Kathmandu;

The CV of the metals in dust samples ranged from 20.3 to 239%. Five metals (Ag, Cd, Cu, Pb, and As) showed relatively higher variability in house dust suggesting the possibility of point source pollution. The Cd had the highest CV with a score of 239%, suggesting a greater variation of Cd in house dust. Fe had the lowest CV with a score of 20.3%, suggesting weak variability and constant distribution across the study area.

3.4. Enrichment factor (EF)

To investigate the degree of enrichment of metals in soil and dust of urban Nepal, EF of individual metal was estimated and is presented in Table 2. The average EF estimated for Ag, Cd, Co, Cr, Cu, Ni, Pb, Sb, Fe, Mn, Zn, and As in soil were 2.52, 2.0, 1.24, 1.90, 3.75, 1.00, 3.82, 2.38, 0.01, 0.96, 6.77, and 0.27, respectively. The average EF of HMs in surface soil were in the order of Zn > Pb > Cu > Ag > Sb > Cd > Cr > Co > Mn > As > Fe. The result showed that the majority of the HMs considerably enriched in surface soil except for Ni, Fe, Mn, and As. Han et al. (2006) suggested two categories of metal enrichment based on EF value; EF $\leq$ 2 suggests deficiency to minimal metal enrichment, while EF $\geq$ 2 suggests the high degree of metals enrichment. In this study, the average EF of Ag, Cd, Cu, Pb, Sb, and Zn were highly enriched metal, while Co, Cr, Ni, Fe, Mn, and As had low EF value.

For dust, the average EF estimated for Ag, Cd, Co, Cr, Cu, Ni, Pb, Sb, Fe, Mn, Zn, and As in dust were 1.60, 3.96, 0.70, 1.39, 4.20, 0.87, 6.51, 2.26, 0.01, 1.0, 7.02, and 0.16, respectively. The estimated EF values of HMs in house dust were in the order of Cd > Zn > Pb > Cu > Sb > Ag > Cr > Mn > Ni > Co > As > Fe (Table 2). The average EF values of Zn (7.02) and Pb (6.51) were greater than
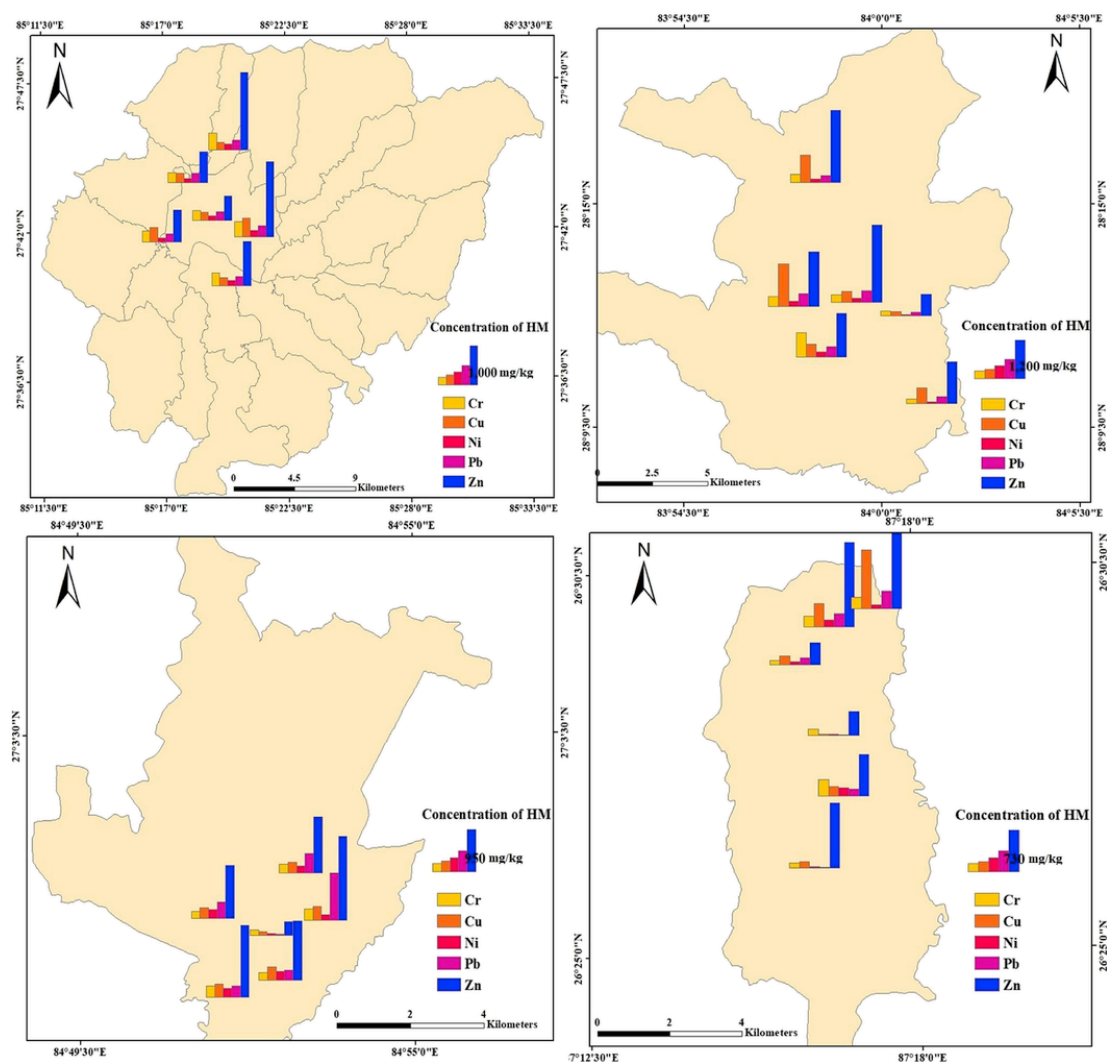
Fig. 4. Spatial maps showing the distribution of priority heavy metals in house dust (mg/kg) from four major cities of Nepal.

Table 2
The calculated enrichment factor (EF) of heavy metals in soil and dust in Nepal.

Metals	Surface soil				House dust			
	Min	Max	Average	SD	Min	Max	Average	SD
Ag	0.79	7.22	2.52	1.64	0.63	4.32	1.60	0.75
Cd	0.96	4.70	2.04	1.06	1.11	92.1	3.96	18.1
Co	0.43	2.90	1.24	0.59	0.20	1.41	0.70	0.31
Cr	0.72	4.78	1.90	1.03	0.51	2.91	1.39	0.63
Cu	1.54	10.4	3.75	2.60	0.41	13.8	4.19	3.51
Ni	0.34	2.56	1.00	0.54	0.21	1.46	0.87	0.36
Pb	1.85	8.81	3.82	1.76	0.51	18.9	6.51	4.13
Sb	1.03	7.07	2.38	1.44	0.68	5.79	2.26	1.27
Fe	0.01	0.03	0.01	0.01	0.01	0.01	0.01	0.00
Mn	1.00	1.00	0.96	0.00	1.00	1.00	1.00	0.00
Zn	2.73	20.2	6.77	4.52	1.24	20.25	7.03	4.06
As	0.05	1.37	0.27	0.29	0.02	0.68	0.17	0.14

5, suggesting household dust in urban Nepal is significantly enriched (Loska et al., 1997; Cheng et al., 2018).

3.5. Sources identification of HMs

To evaluate the extent and sources of metal contamination, a PCA was applied to whole metal data set of soil and dust separately. PCA offers a reduction in the initial dimension of the dataset. Varimax rotation with Kaiser Normalization was used to simplify the coefficient factors. Altogether, four major factors were extracted with an eigenvalue greater than 1 which accounted for 81.7% of the total variance. The results of the PCA together with factor loading obtained after varimax orthogonal rotation of metal data in the soil are presented in Table S7 and Fig. S4. Factor 1 explained 35.3% of total variance in metal data with highest loadings on Cr (0.928), Ni (0.898), Co (0.807), Fe (0.717), Ag (0.635) and Sb (0.632). Cr has both geogenic and anthropogenic sources in the environment (Yadav and Rajamani, 2004; Möller et al., 2005). Cr and Ni are important constituents of traffic emission due to exhaust emission (Johansson et al., 2009). Cr and Ni are mostly found in the same group as they result from similar sources. This statement was supported by the strong linkage between and Ni and Cr ($r=0.907$, $p<0.01$). In urban soil, Ni may also result from Ni-added gasoline, and Ni-containing parts of automobiles (Yassoglou et al., 1987). Co is generally considered as the marker of parent materials (Wang and Zhang and Liu, 2002). Fe and Co in urban soil come from soil parent materials and is related with the crustal source materials (Cass and McRae, 1983; Ostro et al., 2007; Xu et al., 2014; Chen et al., 2016). This was further evidenced by the lower concentration of Fe in soil than background concentration, and positive relationship between Fe with Co ($r=0.703$, $p<0.01$). Sb is a trace element naturally present in the soil. It is also used in the manufacturing of brake linings, antiseptic agents, plastics, flameproofing materials, and as an additive in tire vulcanization process (Shotyk et al., 2006; Steely et al., 2007). Sb in soil could also originate from mining, smelting, pesticide and other Sb-containing materials. However, no any such mining and smelting industries are present in the study area. Hence, Factor 1 may be defined as the combination of both natural and anthropogenic sources.

Factor 2, which has high loadings on Pb (0.794), Cu (0.704), Zn (0.676), and Cd (0.628), accounted for 21.5% of the variance in soil data. Zn, Cu, Cd, and Pb in urban soil could results from the deposition of aerosols particles released from traffic emission, fossil fuel combustion, building construction, and resuspension of road dust (Zheng et al., 2002; Cyrys et al., 2003; Gray et al., 2003). The wear and tear of vehicles on road can also emit Cu, Zn and Cd in urban soil (Martin et al., 1998). However, the concentration Cd is low in the soil in this study, which indicates natural sources of Cd in the soil. In the vehicle tires, Zn is utilized as vulcanization agent in the form of ZnO, Cu is an important component of car lubricant, and leaded gasoline is the major sources of Pb in urban soil (Singh et al., 2015). Zn in urban soil may also originate from the Industrial source, traffic and garbage (Chen et al., 2012). This indicated traffic emission and industrial pollution are an important source of HMs enrichment in Nepal. Hence, Factor 2 is identified as the anthropogenic sources mainly from traffic emission and industrial source. Factor 3 explored 15.0% of total variance with moderate loadings on As (0.582), Cd (−0.548) and Mn (−0.525). The concentration of As and Cd were not high in this study area and Mn had EF value close to 1, which point to the natural source (Han et al., 2006). Hence these metals especially As and Mn are related to crustal material sources (Ostro et al., 2007). Factor 4 explained 9.94% of total variance in the metal dataset with moderate loadings on As (0.657) and Mn (0.622). Factor 4 had a similar distri-

bution to that of factor 3. Hence, Factors 3 and 4 are referred to as the natural source. Therefore, the distribution of Cr, Ni, Sb, Pb, Cu, and Zn in the urban soil of Nepal is mainly affected by anthropogenic sources, while Cd, Co, Fe, As, and Mn was mainly from natural sources.

For dust, a total of three major factors were extracted with an eigenvalue greater than 1 and accounted for 69.6% of total variance (Table S8 and Fig. S5). Factor 1 explained 41.1% of variation in metal data with highest loadings on Sb (0.868), Ni (0.730), Cr (0.707), Co (0.700), Fe (0.709), Zn (0.671), Pb (0.655), Ag (0.666), and Mn (0.590). Leaching of Cr and Zn from wood preservative used in Nepalese household furniture could be the source of Cr and Zn. Likewise, Cr and Ni are also common constituents of stainless steel (Yoshinaga et al., 2014). Burning of mosquito coils and stainless steel debris in domestic houses is one of the significant sources of Ni (Yoshinaga et al., 2014). Zn, Pb, and Ni in house dust could also result from indoor smoking (Pourkhabbaz and Pourkhabbaz, 2012; Cheng et al., 2018). Smoking of cigarette is the important sources of many metals (especially Pb and Zn) in household dust (Bohlandt et al., 2012). Cobalt in house dust results from the deposition of atmospheric particles generated due to weathering of rocks/soil, volcanic eruption, and forest fire particles. Fe and Cr could be generated from solid waste incineration (Yoo et al., 2002). Fe can also result from a metallic substance present in the household. However, Fe concentration in dust in this study was lower than background concentration suggesting the crustal material source (Ostro et al., 2007). Traffic emission is the significant source of Pb in urban dust (Amato et al., 2014; Chen et al., 2014). Lead-based petrol is widely used as fuel in Nepal. Pb deposited in the soil may go into the urban house dust through resuspension (Cheng et al., 2014). Nepal is one of the least developed countries in the world where infrastructural development is limited. In the majority of Nepalese cities, roads are unpaved and not well maintained which results in the generation of dust. Pb shares a common mechanism of house dust contamination with Sb. This was supported by positive correlation between Pb and Sb ($r=0.643$, $p<0.01$). Pb and Sb have been recognized as significant constituents of various toys made of plastics, and metal-based cheap jewelry (Guney and Zagury, 2013; Korfali et al., 2013). Hence, Factor 1 is referred to as a combination of anthropogenic (from domestic household materials, waste incineration, and traffic emissions) as well as natural sources.

Factor 2 accounted for 18.7% of total variance in data, and were moderately loaded with Cd (0.614), Cu (0.677), and As (0.637). Wall paint in domestic houses is important sources of Cu (Tong and Lam, 2000; Chattopadhyay et al., 2003). Another possible source of Cu in house dust could be from airborne particulate (traffic emission and paint additive) related because the abundance of these metals in atmospheric particles is inter-related (Yoshinga et al., 2014). The concentration of Cd and As are low in this study, suggesting natural origin. Therefore, Factor 2 is identified as partly derived from the anthropogenic source and partly from the natural source. PC 3 explained 9.83% of total variance in metal data but didn't contain any significant loading. Hence, the distribution of Sb, Ni, Cr, Zn, Pb and Cu in dust are mostly influenced by anthropogenic source, while Cd, Co, Fe, and As were from the natural source.

3.6. Correlation of HMs with total organic carbon (TOC) and black carbon(BC)

Correlation analysis is a valuable tool for studying the likenesses of paired data and is broadly utilized in HMs data analyses (Bradford et al., 1996). Organic matter has been reported to impact metal ab-

sorption in soils because of cation exchange capacity of organic material (Romic and Romic, 2003). In this study, Pearson's correlation coefficient analysis among HM concentrations, TOC and BC content in soil and dust was applied and are given in Table S9 and S10. Total organic carbon (TOC) and black carbon (BC) have been reported to be good adsorbent for HMs due to their cation exchange capacity, high surface-to-volume ratio and strong affinity (Romic and Romic, 2003; Ray et al., 2012; Wang et al., 2012; Hany et al., 2014). The organic carbon content in soil and dust samples were acquired from our previous studies (Yadav et al., 2016, 2018a). Both TOC and BC content in soil was either not correlated or weakly correlated with HMs. Zn was only moderately correlated with TOC ($r = 0.530$, $p < 0.01$). Like soil, HMs in dust was either not related or weakly related with TOC and BC. TOC was weakly and negatively correlated with Fe ($r = -0.448$, $p < 0.01$). BC in dust was moderately and negatively correlated with Co ($r = -0.514$, $p < 0.01$), and Zn ($r = -0.468$, $p < 0.01$). This weak correlation of HMs irrespective of environmental matrices indicated that soil/dust TOC and BC have little or no influence on HMs.

The inter-relationship of metals can provide significant information about their sources and pathways in the environment (Rodríguez et al., 2008). The Pearson's correlation coefficient was conducted among 12 metals in soil and dust (Table S9 and S10). In soil, Ni was strongly and positively related with Cr ($r = 0.907$, $p < 0.01$), Co ($r = 0.848$, $p < 0.01$) and Cu ($r = 0.502$, $p < 0.01$), indicating similar source of origin. Fe was significantly correlated with Co ($r = 0.703$, $p < 0.01$), Cr ($r = 0.713$, $p < 0.01$) and Sb ($r = 0.594$, $p < 0.01$). Co is abundantly scattered in earth crust and its correlation with Fe indicate their occurrence in surface soil from natural origin. Likewise, Pb was strongly linked with Cu ($r = 0.618$, $p < 0.01$) and Cd ($r = 0.634$, $p < 0.01$). The concentration of As in soil was never correlated with any of metals contents, suggesting its quite different source from other metals. For dust, Sb was significantly correlated with Cd ($r = 0.704$, $p < 0.01$), Co ($r = 0.616$, $p < 0.01$), Pb ($r = 0.643$, $p < 0.01$),

Cr ($r = 0.526$, $p < 0.01$), Cu ($r = 0.584$, $p < 0.01$), and Ni ($r = 0.513$, $p < 0.01$). Mn was positively correlated with Ni ($r = 0.574$, $p < 0.01$), and Fe ($r = 0.726$, $p < 0.01$). Fe and Mn occur abundantly in earth crust thereby rejecting the possibility of anthropogenic influence. Zn was significantly linked with Pb ($r = 0.524$, $p < 0.01$) and Sb ($r = 0.573$, $p < 0.01$). The significant correlation among Zn, Pb, and Sb indicates their common origin especially from the anthropogenic source which is in agreement with PCA results.

3.7. Health risk assessment

Seven priority HMs (As, Cd, Cr, Cu, Ni, Pb, and Zn) were included for estimating the health risk because of their strong toxicity potential to humans (USEPA, 2016). The non-carcinogenic and carcinogenic health risks of these seven HMs were estimated for adult (18–70 yrs) and children (1–17 yrs) through three different pathway of intake (i.e. ingestion, inhalation, and dermal contact). The human health risk of HMs in soil and dust was estimated separately and are presented in Tables 3 and 4. The result showed that ingestion was the major pathway of HMs exposure to the local population both in soil and dust environment, followed by dermal contact. This is consistent with previous studies reported worldwide (Wei et al., 2015; Mehr et al., 2017; Devi and Yadav, 2018). Comparatively, children were more vulnerable to HMs exposure than the adult. Only, Cr showed higher HI value (>1) for both adult and children in soil (1.78 and 4.59, respectively), while Cr and Pb had higher HI values for both adult and children in the dust (1.39, 1.01, 4.04, and 1.7, respectively). This suggests a significant non-carcinogenic risk of Cr and Pb for adult and children in this study area. The carcinogenic risk (CR) estimated for adult and children in soil and dust were close to each other. The average CR values for adult and children in soil (1.07×10^{-3} and 1.58×10^{-3}, respectively) and dust (1.21×10^{-3}, 1.78×10^{-3}, respectively) exceeded the acceptable level of carcinogenic risk for human (1×10^{-6}), suggesting HMs pollution in this area can pose a car-

Table 3
Non-carcinogenic risks and carcinogenic risk of seven priority heavy metals in soil.

Adults		Cd	Cr	Cu	Ni	Pb	Zn	As
HQ ingestion	Kathmandu	4.33E-03	3.49E-01	2.66E-02	2.00E-02	1.56E-01	1.18E-02	2.68E-02
	Pokhara	3.76E-03	3.26E-01	3.19E-02	1.95E-02	1.48E-01	1.99E-02	1.48E-01
	Birgunj	4.77E-03	3.15E-01	2.02E-02	1.69E-02	1.30E-01	1.15E-02	5.80E-02
	Biratnagar	3.05E-03	3.70E-01	2.20E-02	2.14E-02	1.27E-01	1.05E-02	5.20E-02
HQ inhalation	Kathmandu	1.27E-06	1.08E-02	7.78E-06	5.71E-06	4.57E-05	3.48E-06	–
	Pokhara	1.11E-06	1.01E-02	9.32E-06	5.56E-06	4.33E-05	5.84E-06	–
	Birgunj	1.40E-06	9.73E-03	5.90E-06	4.84E-06	3.81E-05	3.37E-06	–
	Biratnagar	8.97E-07	1.14E-02	6.43E-06	6.12E-06	3.71E-05	3.10E-06	–
HQ dermal	Kathmandu	4.79E-04	9.64E-02	4.90E-04	4.10E-04	5.70E-03	3.27E-04	–
	Pokhara	4.16E-04	9.02E-02	5.87E-04	3.99E-04	5.40E-03	5.49E-04	–
	Birgunj	5.27E-04	8.72E-02	3.72E-04	3.47E-04	4.76E-03	3.17E-04	–
	Biratnagar	3.37E-04	1.02E-01	4.05E-04	4.39E-04	4.63E-03	2.91E-04	–
Hazard index (HI)		1.77E-02	1.78	1.02E-01	7.95E-02	5.82E-01	5.52E-02	2.84E-01
Cancer Risk (CR)		6.00E-05	5.13E-04	–	3.29E-04	1.38E-04	–	3.25E-05
Children								
HQ ingestion	Kathmandu	4.33E-03	3.49E-01	2.66E-02	2.00E-02	1.56E-01	1.18E-02	2.68E-02
	Pokhara	3.76E-03	3.26E-01	3.19E-02	1.95E-02	1.48E-01	1.99E-02	1.48E-01
	Birgunj	4.77E-03	3.15E-01	2.02E-02	1.69E-02	1.30E-01	1.15E-02	5.80E-02
	Biratnagar	3.05E-03	3.70E-01	2.20E-02	2.14E-02	1.27E-01	1.05E-02	5.20E-02
HQ inhalation	Kathmandu	2.08E-06	1.76E-02	1.27E-05	9.34E-06	7.46E-05	5.69E-06	–
	Pokhara	1.81E-06	1.64E-02	1.52E-05	9.09E-06	7.07E-05	9.55E-06	–
	Birgunj	2.29E-06	1.59E-02	9.65E-06	7.91E-06	6.23E-05	5.51E-06	–
	Biratnagar	1.47E-06	1.87E-02	1.05E-05	1.00E-05	6.06E-05	5.06E-06	–
HQ dermal	Kathmandu	4.02E-03	8.10E-01	4.12E-03	3.44E-03	4.79E-02	2.75E-03	–
	Pokhara	3.50E-03	7.57E-01	4.93E-03	3.35E-03	4.54E-02	4.61E-03	–
	Birgunj	4.43E-03	7.32E-01	3.12E-03	2.91E-03	4.00E-02	2.66E-03	–
	Biratnagar	2.83E-03	8.59E-01	3.40E-03	3.68E-03	3.89E-02	2.44E-03	–
Hazard index (HI)		3.07E-02	4.59	1.16E-01	9.13E-02	7.34E-01	6.62E-02	2.84E-01
Cancer Risk (CR)		8.83E-05	7.55E-04	–	4.84E-04	2.04E-04	–	4.97E-05

Table 4
Non-carcinogenic risks and carcinogenic risk of seven priority heavy metals in dust.

Adults		Cd	Cr	Cu	Ni	Pb	Zn	As
HQ ingestion	Kathmandu	7.05E-03	3.59E-01	2.35E-02	2.13E-02	2.25E-01	1.34E-02	2.15E-02
	Pokhara	3.50E-02	3.17E-01	4.59E-02	1.72E-02	2.43E-01	1.69E-02	6.80E-02
	Birgunj	1.19E-02	2.13E-01	2.05E-02	2.57E-02	3.79E-01	1.40E-02	4.09E-02
	Biratnagar	6.70E-03	1.72E-01	2.56E-02	1.18E-02	1.27E-01	1.00E-02	3.20E-02
HQ inhalation	Kathmandu	2.07E-06	1.11E-02	6.87E-06	6.08E-06	6.58E-05	3.93E-06	–
	Pokhara	1.03E-05	9.78E-03	1.34E-05	4.92E-06	7.11E-05	4.97E-06	–
	Birgunj	3.49E-06	6.59E-03	5.99E-06	7.33E-06	1.11E-04	4.11E-06	–
	Biratnagar	1.97E-06	5.32E-03	7.49E-06	3.38E-06	3.70E-05	2.95E-06	–
HQ dermal	Kathmandu	7.80E-04	9.92E-02	4.33E-04	4.36E-04	8.22E-03	3.69E-04	–
	Pokhara	3.87E-03	8.77E-02	8.46E-04	3.53E-04	8.88E-03	4.67E-04	–
	Birgunj	1.31E-03	5.90E-02	3.77E-04	5.26E-04	1.39E-02	3.86E-04	–
	Biratnagar	7.40E-04	4.77E-02	4.72E-04	2.42E-04	4.63E-03	2.78E-04	–
Hazard index (HI)		6.74E-02	1.39	1.18E-01	7.76E-02	1.01	5.58E-02	1.62E-01
Cancer Risk (CR)		2.29E-04	4.00E-04	–	3.21E-04	2.40E-04	–	1.86E-05
Children								
HQ ingestion	Kathmandu	1.01E-02	5.14E-01	3.37E-02	3.05E-02	3.23E-01	1.92E-02	3.09E-02
	Pokhara	5.02E-02	4.55E-01	6.58E-02	2.47E-02	3.48E-01	2.42E-02	9.75E-02
	Birgunj	1.70E-02	3.06E-01	2.93E-02	3.68E-02	5.44E-01	2.00E-02	5.86E-02
	Biratnagar	9.60E-03	2.47E-01	3.67E-02	1.70E-02	1.82E-01	1.44E-02	4.58E-02
HQ inhalation	Kathmandu	3.39E-06	1.81E-02	1.12E-05	9.93E-06	1.08E-04	6.42E-06	–
	Pokhara	1.68E-05	1.60E-02	2.20E-05	8.04E-06	1.16E-04	8.12E-06	–
	Birgunj	5.70E-06	1.08E-02	9.79E-06	1.20E-05	1.81E-04	6.71E-06	–
	Biratnagar	3.22E-06	8.69E-03	1.22E-05	5.53E-06	6.05E-05	4.82E-06	–
HQ dermal	Kathmandu	6.55E-03	8.33E-01	3.63E-03	3.66E-03	6.90E-02	3.10E-03	–
	Pokhara	3.25E-02	7.36E-01	7.10E-03	2.96E-03	7.45E-02	3.92E-03	–
	Birgunj	1.10E-02	4.96E-01	3.17E-03	4.42E-03	1.16E-01	3.24E-03	–
	Biratnagar	6.22E-03	4.00E-01	3.96E-03	2.04E-03	3.88E-02	2.33E-03	–
Hazard index (HI)		1.43E-01	4.04	1.83E-01	1.22E-01	1.70	9.04E-02	2.33E-01
Cancer Risk (CR)		3.37E-04	5.89E-04	–	4.73E-04	3.53E-04	–	2.84E-05

cinogenic risk to the local population. Among HMs, Ni and Pb in soil showed a relatively higher cancer risk than other metals, while Cd and Pb were potential metal causing cancer risk to the local population in the dust. In summary, it is inferred that Cr and Pb are the main contaminants that posed a non-carcinogenic risk to the human population, while Ni, Cd, and Pb showed higher carcinogenic risk to both adult and children.

4. Conclusions

In this study, the spatial distribution and sources of metals were investigated in surface soil and house dust from four major urban areas of Nepal. Generally, the concentration of $\sum_7$HMs measured in dust was slightly higher than those in the soil. The median concentration of Ag, Cd, Co, Cr, Cu, Ni, Pb, Sb, Mn and Zn in soil and dust were 2–13 times greater than background concentration in shale, suggesting either contamination or influence of pedogenic factors. Zn was the most prominent metal identified in dust and soil. The result of EF showed that the majority of the HMs is considerably enriched in surface soil except for Ni, Fe, Mn, and As. The average EF estimated for Zn, Cd, and Pb greater than 5, suggested significant enrichment in the dust. PCA analysis showed that Cr, Ni, Sb, Ag, Pb, Cu, and Zn in surface soil are mainly affected by anthropogenic source, while Co, Fe, As, Mn and Cd were from natural sources. The poor correlation of HMs with TOC and BC both in soil and dust suggested little or no influence on HMs contamination. Health risk assessment indicated ingestion as the major pathways of HMs exposure to the human population. Children were more prone to HMs exposure than the adult. There were significant non-carcinogenic and carcinogenic risks from HMs in soil and dust in Nepal. Cr and Pb were the main contaminants that posed a non-carcinogenic risk to the human population, while Ni, Cd, and Pb showed higher carcinogenic risk to both adult and children. Hence, assessment of HMs in multi-environmen-

tal matrices of other urban areas of Nepal should be investigated to protect soils from long-term heavy metal accumulation.

Appendix A. Supplementary data

Supplementary data related to this article can be found at https://doi.org/10.1016/j.chemosphere.2018.11.202.

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