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Dust–Metal Sources in an Urbanized Arid Zone: Implications for Health-Risk Assessments

Leticia García-Rico[1,2] · Diana Meza-Figueroa[3] · A. Jay Gandolfi[4] · Rafael Del Río-Salas[5] · Francisco M. Romero[6] · Maria Mercedes Meza-Montenegro[7]

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Abstract The available information concerning metal pollution in different dust sources and the health effects in children remains limited in Mexico. This study focuses on Hermosillo, which is an urbanized area located in the Sonoran Desert in which soil resuspension and dust emission processes are common. The metal content of arsenic (As), chromium (Cr), manganese (Mn), and lead (Pb) were determined in three dust sources (playgrounds, roofs, and roads), each representing different exposure media (EM) for these elements. The metal levels in dust were found in the order of Mn > Cr > Pb > As with the highest metal content found in road dust. Despite the similar average metal distributions, principal component analysis shows a clear separation of the three EM with playground dust related to Cr and Mn and road dust to As and Pb. However, the geoaccumulation index results indicate that dust samples are uncontaminated to moderately polluted, except for Pb in road dust, which is considerably high. In addition, the enrichment factor suggests an anthropogenic origin for all of the studied metals except for Mn. In this context, the hazard index (HI) for noncarcinogenic risk is >1 in this population and thus represents a potential health risk. The spatial distribution for each metal on EM and the HI related to the marginality index could represent a more accurate decision-making tool in risk assessment studies.

✉ Maria Mercedes Meza-Montenegro
mmeza@itson.edu.mx

[1] Centro de Investigación en Alimentación y Desarrollo, A.C., km 0.6 carretera a la Victoria, Hermosillo, Sonora, Mexico

[2] Programa de Doctorado en Ciencias Especialidad en Biotecnología, Instituto Tecnológico de Sonora, 5 de Febrero 818 Sur, 85000 Cd. Obregón, Sonora, Mexico

[3] División de Ciencias Exactas y Naturales, Departamento de Geología, Universidad de Sonora, Rosales y Encinas, 83000 Hermosillo, Sonora, Mexico

[4] Department of Pharmacology and Toxicology, College of Pharmacy, University of Arizona, 1295 N. Martin, 85721 Tucson, AZ, USA

[5] Estación Regional del Noroeste, Instituto de Geología, Universidad Nacional Autónoma de México, Colosio y Madrid s/n, 83240 Hermosillo, Sonora, Mexico

[6] Instituto de Geología, Universidad Nacional Autónoma de México, Ciudad Universitaria, Delegación Coyoacán, 04510 México, D.F., Mexico

[7] Departamento de Recursos Naturales, Instituto Tecnológico de Sonora, 5 de Febrero 818 Sur, 85000 Cd. Obregón, Sonora, Mexico

Processes linked to climate change are affecting urban air quality in arid climate zones. These processes are progressively enhancing soil degradation and the resuspension of dust in the atmosphere (Al-Khashman 2013; Del Rio-Salas et al. 2012; Habil et al. 2013). Economic and industrial activities associated with rapid urbanization may increase the metal contents in urban dust, thus presenting a health risk to humans (Amato et al. 2011; Charlesworth et al. 2011; Chen et al. 2014; Habil et al. 2013). In arid zones, where resuspension and subsequent sedimentation processes can be significant, soil pollution represents not only an important dust source to the atmosphere but also a medium for pollution exposure to humans, particularly children, who are more susceptible than adults to the adverse effects of soil ingestion (Elom et al. 2013; Gamiño-Gutiérrez et al. 2013; Guney et al. 2010). Urban dust sources commonly include road-, playground-, and soil-derived dust. Resuspension of urban dust enhances the transport and further distribution of metals, thus affecting the urban environment quality and human health (Charlesworth et al. 2011).

Several studies related to human health-risk assessment of metals focused on soil ingestion (Hu et al. 2011; Cai et al. 2013; Rout et al. 2013; Xu et al. 2013) and the ingestion of road dust (Amato 2011; Chang et al. 2009; Shi et al. 2008). Soils from playgrounds are reservoirs of metals from a variety of sources including vehicle emissions, combustion, and industrial wastes (Charlesworth et al. 2011; Wei and Yang 2010). Metal exposure from soil is likely to occur in children through playgrounds of schools in urban cities. The risk could be increased if these sensitive areas are located near busy traffic because metal contents in road dust are reported to be higher than are those in urban soils in areas that are located near busy traffic because metal contents in road dust are reported to be higher than are those in urban soils (Chang et al. 2009; Shi et al. 2008).

The exposure risk is especially high for children because they have a higher susceptibility to environmental pollution (Chen et al. 2014; Gamiño-Gutiérrez et al. 2013). Children spend more time in outdoor activities; therefore, they are more exposed to metals by ingestion through hand-to-mouth pathways (Charlesworth et al. 2011; Habil et al. 2013; Okorie et al. 2012). Airborne dust is an important transport mechanism for metals in arid zones (Charlesworth et al. 2011; Del Rio-Salas et al. 2012). Roofs are interesting sampling sites because the suspended dust could provide information about atmospheric pollution during the dry season when the total suspended particulate matter can reach maximum values (Moreno-Rodríguez et al. 2015).

Certain studies have reported associations between metal exposure from dust sources and lung diseases in adults. However, the risk assessment is typically estimated by considering particulate matter (PM_{10}) or soil- and road-dust ingestion (Kong et al. 2011b; Kesavachandran et al. 2013). In this context, it is important to conduct studies in children looking for associations including soil resuspension. Del Rio-Salas et al. (2012) reported that dust emission related to soil resuspension as a major contributor to the atmospheric pollution in arid zones.

The assessment of human health risks from environmental pollution is necessary to regulate metal concentrations in possible sources of pollutants. Geographic Information System (GIS)—based maps are useful to show the spatial distribution of pollutants, socioeconomic conditions, vulnerability (marginality index), and number of people at risk for excessive intake. Published studies on mapping human health risk are related to exposure to metals in soil, water, and food (Rodríguez-Salazar et al. 2011; Dao et al. 2013); however, to our knowledge, no studies have been published on mapping human health risks related to resuspended dust in urban environments including settled dust on roofs.

Therefore, the aim of this study is to (1) evaluate arsenic (As), chromium (Cr), lead (Pb), and manganese (Mn) levels in dust from three different EM (playground, road, roof/airborne); (2) evaluate the health risk exposure from the three sources; and (3) determine the spatial distribution of hazard indices (HIs) associated with the marginality index within the city.

Materials and methods

Study Area

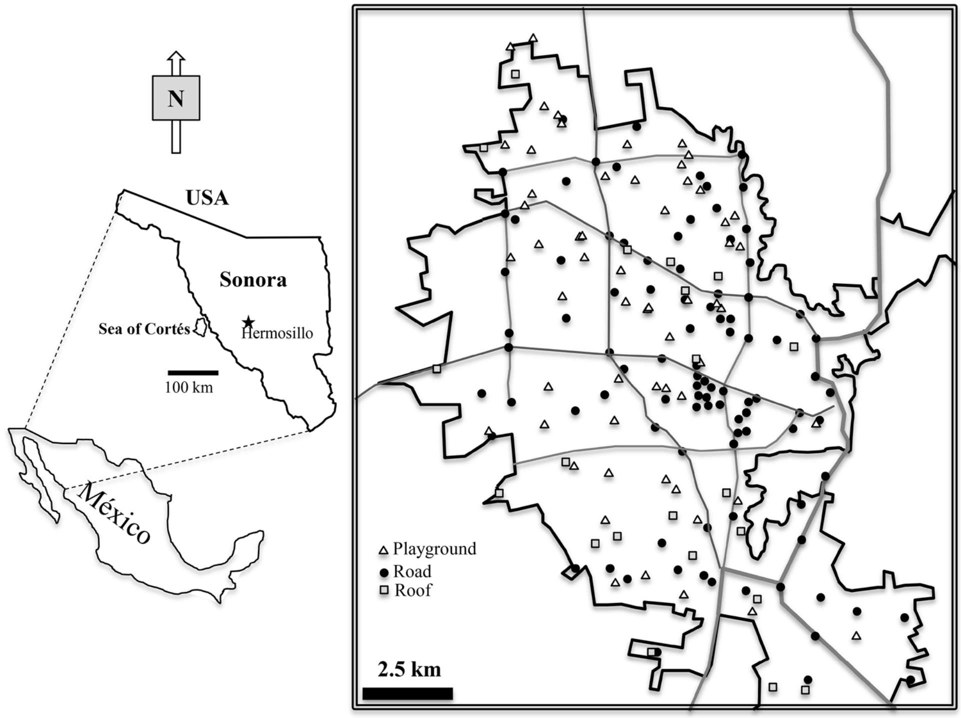
Hermosillo is located in northwestern Mexico within the Sonoran Desert, which extends to Arizona and California in the United States and represents one of the most important arid zones in the world (Fig. 1). The weather conditions are typical of desert environment, with temperatures ranging from 35 to 49 °C during summer and 5–8 °C during winter. The climate is dry most of the year, and the region is characterized by arid to semiarid conditions with low annual precipitation (75–300 mm).

Over the last two decades, the city has experienced rapid growth from 406,417 inhabitants in 1990 to 715,061 in 2010 (INEGI 2015) with a population growth rate of approximately 3 %, which is greater than the state population growth rate of approximately 1.4 %. The main economic activities in the region are industrial, manufacturing, cattle farming, agriculture, and mining. The rapid growth of the city, coupled with a lack of planning for urban infrastructure, has resulted in negative consequences for the air quality.

Sample Collection

Dust samples (220) were collected for this study, thus ensuring a representative sampling within the urban area. The purpose was to obtain a set of samples across industrial, commercial, residential, and high and low traffic-density areas including different socioeconomic neighborhoods. As described by Del Rio-Salas et al. (2012), 116 urban street dust samples were obtained. Briefly, settler dust at each sampling site was collected from within 2 m^2 using a polyethylene brush, a tray, and containers. In addition, 79 dust samples were obtained from playgrounds, and 25 deposited dust samples were obtained from elementary school roofs. The collected dust samples (200 g) were dried at 35 °C, sieved (325 and −325 mesh), and stored at room temperature until analysis. The 325-mesh fraction corresponds to particle sizes between 74 and

Fig. 1 Location of study area and sampling sites. The *gray lines* are the main roads



44 µm, whereas the −325 mesh fraction corresponds to particle sizes <44 µm. The fine-grained fraction (−325 mesh) is most likely soil that was resuspended by wind and anthropogenic activities (Laidlaw and Filippelli 2008) and represents a fraction of the potentially inhaled and ingested dust.

Total Metal Content in Urban Dust

As, Cr, Pb, and Mn concentrations in fine-grained fractions (<44 µm) from playground-, roof-, and road-dust samples were analyzed in duplicate using a portable X-ray fluorescence analyzer (Niton XL3t Analyzer, Thermo Scientific, Inc., MA, USA) according to method 6200 from the United States Environmental Protection Agency (USEPA). Field-portable X-ray fluorescence spectrometry was used for the determination of elemental concentrations in the dust samples. The detection limits for As, Cr, Mn, and Pb were 7, 25, 28, and 13 mg kg^{-1}, respectively. To validate the quality of the acquired data, a subset of the samples was analyzed at the ALS CHEMEX Laboratory in Toronto, Canada. Compared with the present measurements, the obtained results showed a 95 % confidence level, which is highly acceptable.

Assessment of Contamination Level

The geoaccumulation index (Igeo) method has been widely used to evaluate metal pollution levels in soils and street dust (Kong et al. 2011a) with a few studies focused on roof dust. The Igeo is calculated using the following equation:

$$I_{\text{geo}} = \text{Log}_2[C_n/1.5B_n] \quad (1)$$

where C_n represents the measured concentration of metal *n*, and B_n is the geochemical background value of metal *n*. The constant 1.5 is used to compensate for natural fluctuations of the studied metal in the environment and to detect small anthropogenic influences (Wei and Yang 2010). In this study, B_n is the value of metal n in the local geochemical background. The Igeo value is classified as follows: uncontaminated (Igeo ≤ 0), uncontaminated to moderately contaminated ($0 <$ Igeo ≤ 1), moderately contaminated ($1 <$ Igeo ≤ 2), moderately to heavily contaminated ($2 <$ Igeo ≤ 3), heavily contaminated ($3 <$ Igeo ≤ 4), heavily to extremely contaminated ($4 <$ Igeo ≤ 5), and extremely contaminated (Igeo ≥ 5).

To test whether the metals have natural origins or are derived from anthropogenic activity (or both), the enrichment factor (Ef) was calculated. In this study, the *Ef* was calculated according to Chen et al. (2014) as follows:

$$Ef = (M_1/\text{Eref}_1)_{\text{sample}} \Big/ (M_2/\text{Eref}_2)_{\text{crust}} \quad (2)$$

M_1 is the concentration of metal in the analyzed sample, and Eref_1 is a reference element in the analyzed sample (iron [Fe] in this study), M_2 is the same element as M_1 but in the local geochemical background (based on the analysis of six unpolluted local soils from different parts of the city), and Eref_2 is the reference element in the local geochemical background. Values of *Ef* close to 1 indicate a natural origin, whereas values >1 are considered to have originated primarily from anthropogenic sources (Chen et al. 2014).

Risk Estimation

The health risk assessment model derived from the USEPA (Integrated Risk Information System) was applied to calculate As, Cr, Mn, and Pb intake for dust to estimate noncancer (chronic) and carcinogenic toxic risk exposure for children by way of dust particles (<44 µm fraction) (USEPA 2009, 2013) as follows:

$$CDI_{ingestion} = [C \times IngR \times EF \times ED] / [BW \times AT]\, 10^{-6} \quad (3)$$

CDI represents the chemical daily intake (mg kg^{-1} day^{-1}), and *C* is the mean metal concentration (mg kg^{-1}) in the dust samples. Conservative estimates of dust-ingestion rates (IngR) for children were chosen based on a recommended central tendency of 200 mg day^{-1} for the age group 6 to <21 years including soil and outdoor settled dust (USEPA 2009). The exposure frequency (EF) was 255 days $year^{-1}$ assuming that the average number of days attending school for children in Mexico. ED is the exposure duration (9 years) corresponding to the average age of third-grade students in Mexico. An average body weight (BW) of 15 kg for children was assumed (USEPA 2009, 2013). Averaging time (AT) was estimated as the ED $\times$ 365 days for noncarcinogens and 70 $\times$ 365 days for carcinogens.

The potential noncarcinogenic hazard quotient (HQ) and carcinogenic risks were calculated using the following equations:

$$HQ_{ingestion} = CDI_{ingestion}/RfD \quad (4)$$

$$\text{Carcinogenic risk} = CDI \times SF_0 \quad (5)$$

where HQ is an estimate of the daily exposure to the human population that is likely to be without an appreciable risk of deleterious effects over a lifetime. Thus, $HQ \leq 1$ suggests unlikely adverse health effects, whereas $HQ > 1$ suggests the probability of adverse health effects. The HQ can be added to generate a HI that may allow estimates for the risk of mixing contaminants; therefore, the HQ for each metal at a location was summed to generate the HI. In this study, the slope factor (SF_0) and oral reference doses (RfD) were obtained from the integrated risk information system, and regional screening levels (USEPA 2013) and are listed in Table 3. However, the USEPA has not established an RfD for Pb; therefore, the RfD used in this study was 3.5E–03 mg kg^{-1} day^{-1}, which was calculated from the provisional tolerable weekly Pb intake limit (25 mg kg^{-1} BW^{-1}) recommended by the Food and Agriculture Organization of the United Nations/World Health Organization (Hu et al. 2011). Because of the lack of data for total Cr concentration, the RfD and SF_0 of Cr VI were used. Carcinogenic risk was estimated only for As and Cr. Carcinogenic risk is the probability of an individual developing any type of cancer from lifetime exposure to carcinogenic hazards. The acceptable or tolerable risk for regulatory purposes is over the range of 1E–06 to 1E–04 (USEPA 2013).

Statistical Analyses and Geographic Information System

Descriptive statistical parameters such as minimum, maximum, mean, and SD were estimated for the metal content in the samples studied. In this study, principal component analysis (PCA) was used to decrease the dimensions of the space in which variables are projected and as a qualitative pattern recognition method, which could indicate the sources of metal enrichment in the dust samples studied. The software StatistiXL 1.6 was used in this study. The number of significant PCs used for interpretation is based on the Kaiser criterions, which consists of retaining factors with eigenvalues >1 and explained variance ≥70 % (Mostert et al. 2010). For the assumption of normality and equality of variance of the data, log-transformation was used. The significance of the differences between mean metal concentrations in dust from the sources studied was determined using ANOVA with Bonferroni correction. Statistical analyses were performed with Stata 9.0 Software (College Station, TX, USA, 2007).

All of the samples with reported assays were digitally processed using Geosoft's Target for ArcGIS and ArcGIS 10.1 software (Environmental Systems Research Institute, Inc., ESRI, 2005). Target for ArcGIS is a powerful suite of tools designed for two- and three-dimensional modeling, and ArcGIS is the most widely used software for managing and analyzing spatial and database-linked information such as that from a GIS. Two different interpolation methods, minimum curvature and inverse distance weighted, were used to interpolate the geochemical results. Due to the nature of distributions and distances between roof, road, and playground dust samples, the minimum curvature method was used to interpolate their analytical results. The minimum curvature method is an interpolation method used in most geoscience fields. This method is not an exact interpolator, indicating that the exact original data are not preserved; however, the method generates the smoothest possible surface while considering the original data at each point, thus attempting to apply a minimum amount of bending between data. This method produces a grid from the original data points and fills empty spatial points by repeatedly applying an equation over the grid in an attempt to smooth the grid. Each pass over the grid is counted as a single iteration, and a maximum of 100 iterations were used in the samples discussed herein.

The demographic characteristics of the Hermosillo census (INEGI 2015) related to one of the marginality index indicators were interpolated with the HI generated by GIS. Marginality indicators were generated using the Urban Basic Geostatistical Area (UBGA). The Mexican UBGA is defined as a geographical area occupied by a set of well-defined blocks of streets, avenues, ramblers, or any other easily identified feature in the field and whose land use is primarily residential, industrial, or commercial with at least 2500 inhabitants.

Results and Discussion

Metal Concentrations in Urban Dust

Statistical descriptions of As, Cr, Mn, and Pb concentrations found in the three different EM (playground dust, roof/airborne dust, and road dust) collected from selected sites in Hermosillo are listed in Table 1. According to the average metal concentrations in the different sources, the concentrations decreased in the order of Mn > Cr > Pb > As. There was considerable variation in the metal contents in the samples with Mn (343.1–975.8 mg kg^{-1}) and Pb (15.2–979.9 mg kg^{-1}) having the highest variation followed by Cr (112.7–273.5 mg kg^{-1}) and As (12.3–28.7 mg kg^{-1}). Table 1 shows that the average As and Cr contents were similar in the three different sources but were highly significant ($p < 0.05$) only for As in roof dust and road dust. The average Mn concentration was higher in playground dust (586.4 mg kg^{-1}) and road dust (550.5 mg kg^{-1}; $p < 0.05$) with lower Mn content in roof dust samples (481.2 mg kg^{-1}). Significant variations ($p < 0.05$) were detected in the three different sources. In the case of Pb, the highest value was found in road dust (126.9 mg kg^{-1}) followed by roof dust (70.3 mg kg^{-1}) and playground dust (55.9 mg kg^{-1}) with statistical differences for Pb in the three main sources ($p < 0.05$). These behaviors may indicate different sources for the metals.

Table 1 Minimum, maximum, average (mg kg^{-1}), and SD values of analyzed elements in dust from selected exposure paths: Playground, roof, road

		Playground	Roof	Road
As	Min–max	12.9–25.3	12.9–18	12.3–28.8
	Average	16.4^{a}	15.1^{a,b}	19.3^{a,c}
	SD	3.5	1.5	4.3
Cr	Min–max	123.8–235	128.8–240.3	112.7–273.5
	Average	161.6^{a}	156^{a}	167.9^{a}
	SD	27.3	31.5	26.1
Mn	Min–max	409.2–781.6	366.7–636.2	343.1–975.8
	Mean	586.4^{a}	481.2^{b}	550.5^{c}
	SD	84.6	71.1	100.6
Pb	Min–max	15.2–176.2	29.3–261.6	21.7–979.9
	Average	55.9^{a}	70.3^{a,b}	126.9^{c}
	SD	32.4	56.8	109.2

Different letters by row are significantly different ($p < 0.05$)

Table 2 shows that the average metal concentrations obtained in this study are comparable with those from other cities worldwide (Ng et al. 2003; Hu et al. 2011; Kong et al. 2011b; Chen et al. 2014; Zhang et al. 2013) except for Cr and Pb (Chen et al. 2014; Zhang et al. 2013). Both of the metal concentrations were similar or higher than those reported from industrialized Chinese cities such as Shanghai and Nanjing (Chen et al. 2014; Zhang et al. 2013). This finding may be related to the number of vehicles in Hermosillo, which is estimated at approximately 300,000 (INEGI 2015); this number of vehicles is significantly high considering the 715,000 inhabitants. In contrast, Cr and Pb content in roof dust in this study are higher than those reported by Meza-Figueroa et al. (2007). This enrichment may be related to the increase in the number of vehicles and because city growth has led to the development of residential areas and new streets and avenues.

As and Mn levels were similar to the reported levels in dust from other southern Sonora cities with inhabitants (<300,000), but Cr and Pb were higher in the present study (Meza-Montenegro et al. 2012). These differences may be related to the city size and industrial activities; however, similar metal contents have been reported in top soils in Mexico City (Rodríguez-Salazar et al. 2011), which is the largest city in the country.

Playgrounds are common sites where children are exposed to dust by way of inhalation and ingestion; if located near high traffic areas, playgrounds should be considered unfriendly environments. A comparison of the As and Pb levels in playgrounds from this study and in certain cities worldwide is shown in Fig. 2. The average As concentration from this study was slightly higher than values reported in other cities (De Miguel et al. 2012; Elom et al. 2013; Iribarren et al. 2009; Guney et al. 2010) except for New Orleans (Mielke et al. 2010; Fig. 2). In addition, the average Pb concentration was lower than that in most cities as shown in Fig. 2 (Cai et al. 2013; Dao et al. 2013; Douay et al. 2007; Elom et al. 2013; Hu et al. 2011; Yoshinaga 2012) but was higher than that in cities in Sweden (Elom et al. 2013), Jordan (Al-khashman 2013), Turkey (Guney et al. 2010), and Spain (De Miguel et al. 2012). It has been documented that the As and Pb content

Table 2 Metal concentrations (average) in playground, roof, and road dusts obtained in the present study compared with some values reported from some worldwide cities (mg kg^{-1})

City/country	Population	As	Cr	Mn	Pb	Reference
Playground dust						
Hermosillo, México	715, 061	16.4	161.6	586.4	55.9	This study
Nanjing, China	3.2 million	12.1	92	575	63	Hu et al. (2011)
Hangzhou, China	1,719,000	NR	54	536	95.4	Zhang and Wang (2009)
Hong Kong, China	6.8 million	NR	NR	496.8	248.1	Lam et al. (2005)
Hong Kong, China	6 million	NR	263	518	77.3	Ng et al. (2003)
School roof dust						
Hermosillo, México	715, 016	15.1	156	481.2	70.3	This study
Xi'an, China	6.5 million	13.2	149.2	558.3	180.9	Chen et al. (2014)
Ma'an, Jordan	NR	NR	NR	48	25.1	Al-Khashman (2013)
Dongying, China	1.83 million	19.1	67.4	572.2	168	Kong et al. (2011b)
Hermosillo, México	740, 000	NR	11.1	NR	36.1	Meza-Figueroa et al. (2007)
Road dust						
Hermosillo, México	715,061	19.3	167.9	550.5	126.9	This study
Ma'an, Jordan	NR	NR	NR	56–110	33–105	Al-Khashman (2013)
Guangzhou, China	>10 million	NR	0.23	NR	65.4	Cai et al. (2013)
Huludao, China	>200, 000	33.1	NR	NR	NR	Xu et al. (2013)
Shanghai, China	NR	NR	145	NR	259	Zhang et al. (2013)
C. Obregón, México	NR	NR	27	622	26.2	Meza-Montenegro et al. (2012)
Navojoa, México	NR	NR	25	583	20.5	Meza-Montenegro et al. (2012)
Newcastle, England	NR	6.4	NR	NR	153	Okorie et al. (2012)
Nanjing, China	3.2 million	13.4	126	646	103	Hu et al. 2011
Fushun, China	NR	14	5558	545.8	50.7	Kong et al. (2011a)
Urumqi, China	1.94	NR	54.3	926.6	53.5	Wei and Yang (2010)

NR not reported

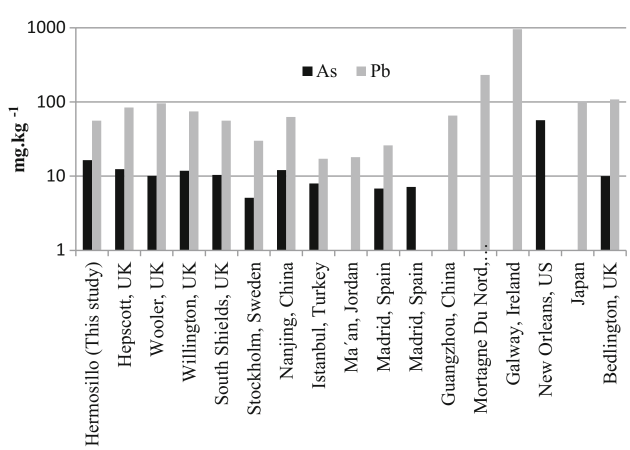


Fig. 2 Comparison of As and Pb levels for playgrounds in cities worldwide (data reported in the literature)

in urban dust could be related to traffic (Amato et al. 2011; Del Rio-Salas et al. 2012; Laidlaw and Filippelli 2008).

The average concentrations of As, Cr, and Pb in the three different EM (Table 2) were higher than the upper continental crust values reported by Rudnick and Gao (2003). The upper crust is the most accessible part of the planet and has historically been the target of geochemical investigations because knowledge of the age and composition of the continental crust is essential for understanding its origin. In this study, As was approximately 2 times higher, whereas Pb was approximately 3, 4, and 7 times higher for playground dust, school roof dust, and road dust, respectively. These high values in the three different sources suggest an anthropogenic source. It is noteworthy that the suspension of road dust during dry seasons (in desert cities) may increase the metal contents in urban areas (Laidlaw and Filippelli 2008).

When the average metal concentrations determined in this study (Table 2) were compared with levels set by the Mexican regulation norms, most of the studied metals did not exceed the permissible values, and only two playground samples and five road dust samples exceeded the recommended levels of As (22 mg kg^{-1}). In the case of Pb, only two samples of road dust yielded values above the recommended limit (400 mg kg^{-1}); both samples correspond to high-traffic areas. According to Del Rio-Salas et al. (2012) and Rodríguez-Salazar et al. (2011), Pb isotopic ratios suggest pollution by Pb in gasoline. In Mexico, using Pb as a gasoline additive has been prohibited since

1987; however, a recent study suggests a mixing of Pb isotopic signatures from leaded gasoline and urban dust. In the case of Mn, which is another potentially risky heavy metal, the Mexican regulations do not yet include a maximum permissible value.

Assessment of Contamination Levels

The metal pollution levels studied in the three sources are shown in Fig. 3. Igeo values in playground, roof, and road dust range from −0.9 to −0.73 for As, 0.4–1.5 for Cr, −1.6 to −0.7 for Mn, and −1.3 to 4.7 for Pb. The average Igeo values in playground, roof, and road dusts decrease in the order of Pb > Cr > As > Mn. The average Igeo values for As in the three different sources were nearly zero; the values were between zero and 1 for Cr, showing uncontaminated to moderately contaminated dust; the values were less than zero for Mn, indicating an uncontaminated environment; and the values were between 1 and 2 for Pb in roof and road dust, which indicates moderate contamination.

To examine the behavior of metals in the three different sources and to consider whether the Ef value is an estimation of the anthropogenic inputs of these metals, metal Ef values were calculated using Fe to normalize the results (Al-Khashman et al. 2013; Chen et al. 2014). The Ef values obtained are shown in Fig. 4. The distribution pattern of the Ef values in the three different sources was Pb > Cr > As > Mn. Slight variations were observed for As (1.7–2), Cr (3.2–4), and Mn (1–1.2); however, Pb yielded Ef values ranging from 3 to 6.5. The highest metal Ef levels were found in roof dust, except for Pb, whose highest value was detected in road dust, whereas the lower Ef values were found in playground dust. The results indicated that Mn is less enriched (Ef ∼ 1) suggesting a natural origin. The Ef values for As, Cr, and Pb in the three different sources were >1, thus indicating an anthropogenic origin most likely related to traffic and industrial emissions. Moreover, in the past, agricultural activities used pesticides such as lead arsenate, chromic oxide, cupric oxide, and arsenic pentoxide (copper chromated arsenate); antifungals and insecticides in wood may reflect an agrarian legacy of contaminant metals.

PCA enables the extraction of three PCs explaining 91.4 % of the total variance (Fig. 5). The first two PCs account for 75.97 % of total variance and explain all of the variables (As, Pb, Mn, and Cr). The PCA plot shows the position of the three sources analyzed (playground, roof, and road), in which the three groups are identified in the two most important factorial plans. The road-dust group is correlated with As and Pb, whereas the playground-dust group is related to Mn and Cr. In this study, Mn was identified as a geogenic element according to the predictions using Igeo and Ef values. Therefore, the geological environment may have been a major contributor to the playground-dust chemistry. Roof-dust samples do not show correlations with the studied metals (Fig. 5). Therefore, the risks are associated with the metal composition of the dust source.

Risk Assessment

A risk assessment was performed based on particles with a size < 44 μm, which may be ingested. The HQs (noncarcinogenic and carcinogenic) for the three different sources

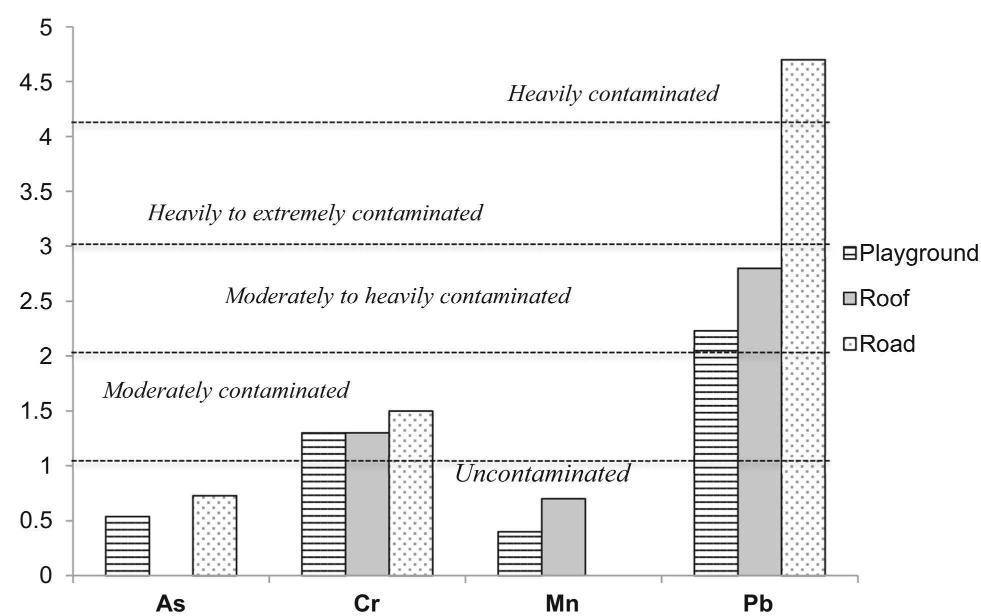
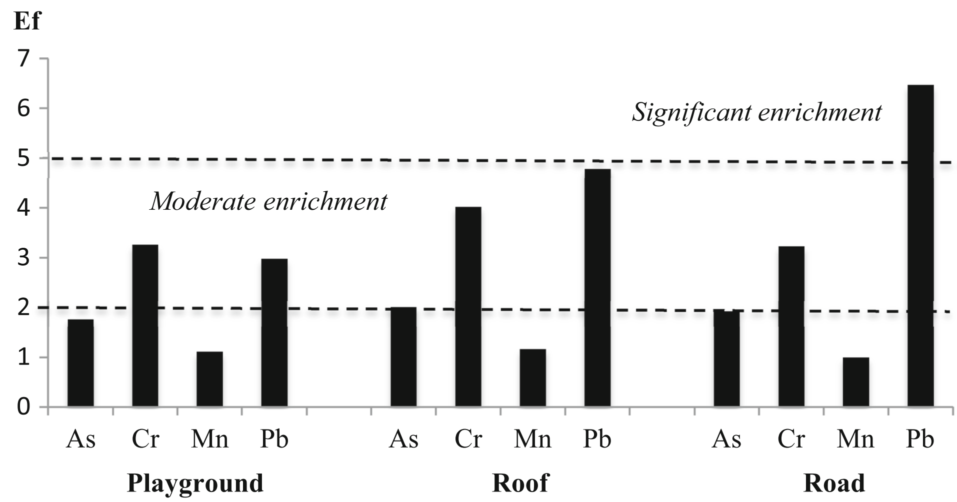


Fig. 3 Geoaccumulation index for the studied exposure path based on minimum, mean, and maximum metal concentrations

Fig. 4 Enrichment factor for studied metals in the studied exposure path (playground, roof, and road). *Dashed line* shows the limit of 1. Values >1 represent anthropogenic sources, and values <1 indicate geogenic sources



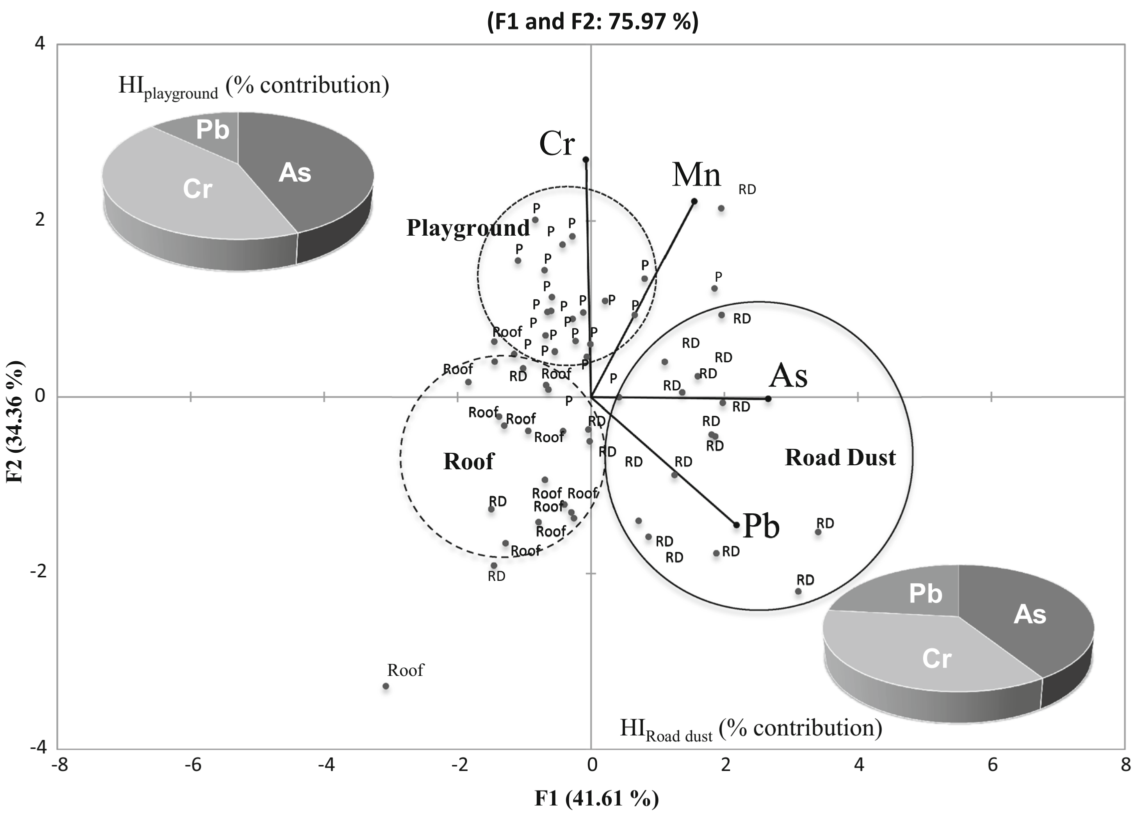


Fig. 5 PCA of the elemental composition of playground, roof, and road dust samples. Data points outside of the highlighted areas (*circles*) do not influence the overall statistical distribution. *RD* road dust, *P* playground dust, *RF* roof dust

are listed in Table 3. For all of the sources, Pb showed the highest HQ values followed by As and Cr. For Mn, the HQ values were similar in the three sources studied. Road-dust samples yielded the highest HQ average values for all of the studied metals. The Pb in road dust (HQ = 2.6) was the only element that exceeded the safe value (HQ = 1), thus representing a potential health risk. When average HQs were compared with the total HQ from each source, the HQs decreased as follows: As ∼ Cr > Pb > Mn. As (39.2–42.5 %) and Cr (34.7–41.7 %) contributed with similar values in the three sources. Meanwhile, Pb was similar in playground dust and roof (12.5–15.8 %) and high in road dust (22.7 %). As for Mn, its contribution was also similar in the study sources but in lowest percentages

Table 3 Risk assessment of metals in children by exposure from playground, roof, and road dusts

Metal		Playground				Roof				Road			
As		C	CDI	HQ	CR	C	CDI	HQ	CR	C	CDI	HQ	CR
RfD = 3E−04	Max	25.3	2E–04	0.79		18	2E–04	0.56		28.8	3E–04	0.89	
SF_0 = 1.5	Min	12.9	1E–04	0.40		12.9	1E–04	0.40		12.3	1E–04	0.38	
Cr	Average	16.4	1E–04	0.51	3E–05	15.1	1E–04	0.47	2.7E–05	19.3	2E–04	0.60	3.5E–05
RfD = 3E–03	Max	235	2.2E–03	0.73		240.3	2.2E–03	0.75		273.5	2.5E–03	0.85	
SF_0 = 5E–01	Min	123.8	1.2E–03	0.38		128.8	1.2E–03	0.40		112.7	1E–03	0.35	
	Average	161.6	6E–04	0.50	9.7E–05	156	1.4E–03	0.48	9.3E–05	167.9	1.6E–03	0.52	1E–04
Mn	Max	781.6	7.3E–03	0.05		636.2	5.9E–03	0.04		975.8	9.1E–03	0.06	
RfD = 1.4E–01	Min	409.2	3.8E–03	0.03		366.7	3.4E–03	0.02		343.1	3.2E–03	0.02	
	Average	586.4	5.5E–03	0.04		481.2	4.5E–03	0.03		550.5	5.1E–03	0.04	
Pb	Max	176.2	1.6E–03	0.47		261.6	2.4E–03	0.70		979.9	9.1E–03	2.61	
RfD = 3.5E–03	Min	15.2	1E–04	0.04		29.3	3E–04	0.08		21.7	2E–04	0.06	
	Average	55.9	5E–04	0.15		70.3	6E–04	0.19		126.9	1E–03	0.34	
HI = $\Sigma HQ_{average}$				1.2				1.2				1.5	

C metal concentration (mg kg^{-1}), *CDI* chemical daily intake (mg kg^{-1} day^{-1}), *CR* carcinogenic risk, SF_0 slope factor (mg kg^{-1} day^{-1})

(2.5–3.5 %). These results highlight the importance of thoroughly exploring the sources of these metals, especially As and Cr.

The HI is obtained when the HQs are added; this index is useful to evaluate the risk by exposure to metal mixtures, and the safe HI value is ≤1. In this study, HI_{total} values ($HQ_{playground\ dust}$ + $HQ_{roof\ dust}$ + $HQ_{road\ dust}$) ranged from 1.2 to 1.5 with the highest contribution to HI from $HQ_{road\ dust}$ (Table 3; Fig. 5). In the three sources studied, Cr and As had a greater contribution to HI_{total} as follows: 84.17 % to $HI_{playground\ dust}$, 79.17 % to $HI_{roof\ dust}$, and 74.67 % to $HI_{road\ dust}$. These results are supported by the PCA (Fig. 5) showing that As and Pb were correlated in road dust, whereas Cr is characteristic of playgrounds. The HI for playgrounds and roofs dusts is 1.2, and the HI for road dust is 1.5 with all of the sources exceeding safe risk values. Therefore, the increased exposure frequency and/or ingestion rate can result in adverse effects to children. Moreover, the HQ and HI values agreed with those found for Igeo and Ef with respect to Cr and Pb, thus showing uncontaminated to moderately contaminated dust from anthropogenic origin.

For the study metals, only for As and Cr exhibit sufficient evidence as carcinogens for humans (USEPA 2009). For this reason, the carcinogenic risk was evaluated only for these elements (Table 3). In this study, Cr exhibited the highest CR values in the three different sources, and road dust showed the highest mean Cr value (1E–04). The As carcinogenic risk was within the range of 2.7E–05 to 3.51E–05 for the three sources. The USEPA (2009) considers that an excess cancer risk is negligible for values lower than 1E–06, whereas values above 1E–04 are sufficiently large such that remediation is desirable. The calculated values in the present study were lower than the safe carcinogenic value. However, is important to note that the carcinogenic levels for Cr in road dust were close to the safe level, thus emphasizing the importance of determining the Cr species. Cr VI is considered carcinogenic, and Cr III is essential for living organisms, but Cr III may be oxidized to Cr VI and enhance its toxicity (Rout et al. 2013). Further studies pertaining to remediation must be performed.

The spatial distributions of HI for the different sources are shown in Fig. 6. HI values above the safe risk level are shown in darkest gray; for playgrounds, the most vulnerable geographic areas were identified considering a high marginality index (soft white shapes, Fig. 6) among the highest HI values. Two main areas (north and south) with HI > 1 were found for roof dust. For road dust, the HI > 1 values were concentrated in the center of the city near high traffic–density roads. Figure 6 shows the importance of a sector-based risk assessment; for example, in the city studied, geographical areas with a concentrated population of children are (1) the peripheries in which irregular population settlements occurred and (2) in the newly developed residential areas, whereas the older population is concentrated in the central area (downtown). Therefore, sensitive populations would be more exposed to metal contaminants depending on their lifestyle. Most of the published GIS-based hazard maps do not consider all of the mentioned variables. Special attention must be focused on certain peripheral areas of the city in which the population presents the highest marginalization index.

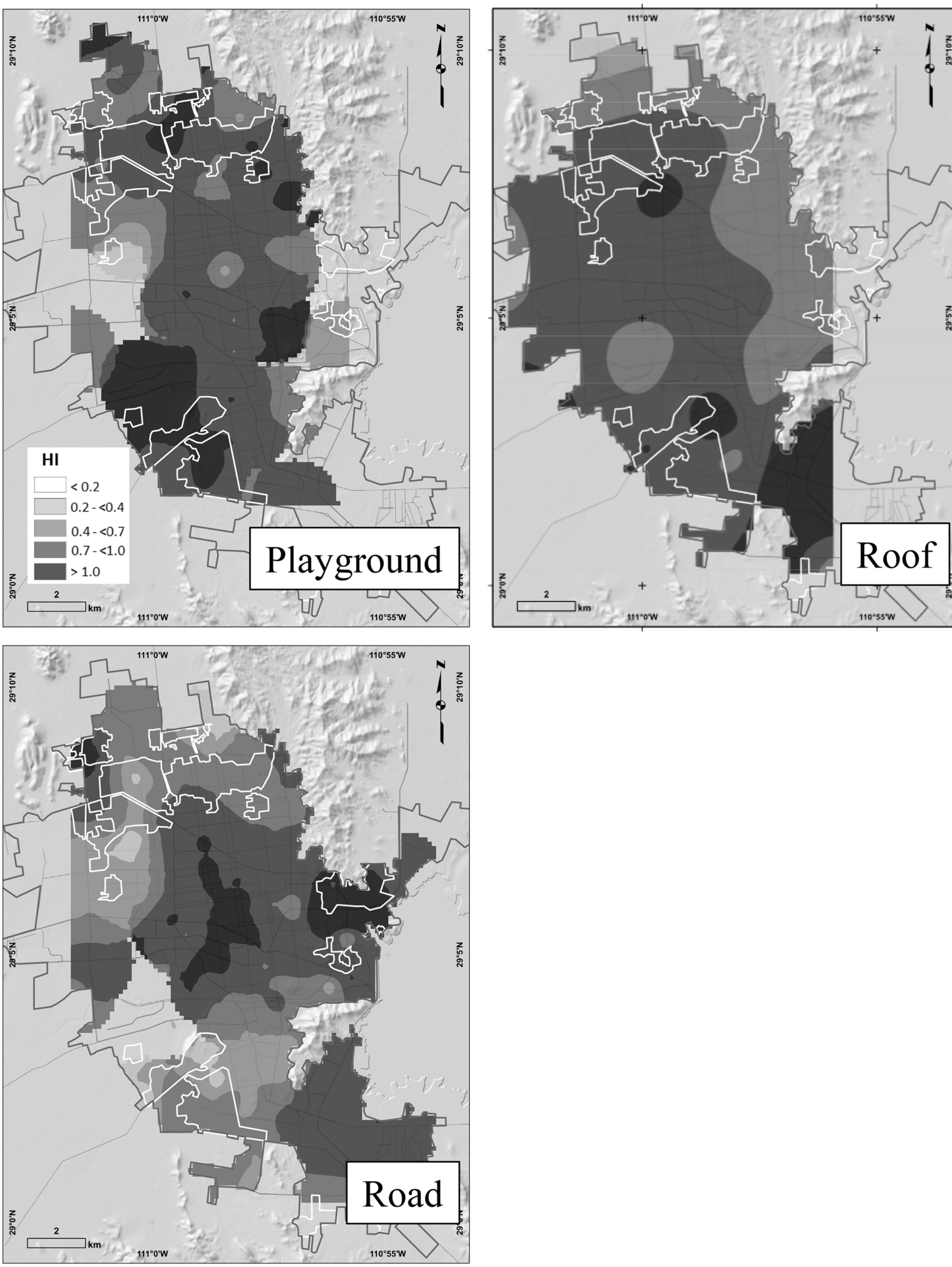


Fig. 6 Interpolated maps of the HI for each studied exposure path (road, playground, and roof) based on As, Pb, Mn, and Cr concentrations. *White lines* represent marginality indexes

The adverse health effects produced by metals in populations with different socioeconomic status exposed to polluted dust are not yet well known primarily because the dust is a transport of multiple contaminants that may act as cofactors and whose effect is unknown (Charlesworth et al. 2011; Moreno-Rodríguez et al. 2015). The HI results from this study (Table 3) were lower than those reported by Xu et al. (2013) for As in street dust in China (HQ values from

0.41 to 2.2; HI values from 0.5 to 2.7) but were similar to the results reported by Hu et al. (2011) for Nanjing street dust (Nanjing is a very large metropolitan city in China).

Conclusions

This study is the first performed in Mexico that estimates the health risk by exposure to multiple metals in three different dust sources and represents an effort to provide a useful tool for decision-making in future biomarker studies. The highest average levels of metals were detected in road dust, except for Mn, which had the highest concentration detected in playground dust followed by road and roof dust samples. Using Igeo and Ef, the metal levels in the studied dust samples are classified as uncontaminated to moderate polluted; all of the studied metals, except for Mn, indicate an anthropogenic origin. PCA analysis shows that road dust was correlated with As and Pb, whereas playground dust was more correlated with Cr and Mn. The HI spatial distribution shows different behaviors for the studied sources. The risk of adverse effects from excess metal intake from dust appears to be high (HI values from 1.2 to 1.5) and represents a potential risk to human health. HI values and marginality index maps could be useful tool in sampling design in exposure biomarkers research, the potential of which has not been explored.

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