Status, source, ecological and health risk assessment of toxic metals and polycyclic aromatic hydrocarbons (PAHs) in street dust of Abadan, Iran

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Abstract

The potential hazard to human health from exposure to toxic metals and polycyclic aromatic hydrocarbons (PAHs) in street dust was assessed using thirty street dust samples collected from Abadan, an industrial city in Southwest Iran. The mean concentration of Pb, Zn, Cu, Cr, Cd, Ni, V, As and Co in the street dust was: 59, 288, 113, 50, 0.5, 57, 36, 7.1 and 7.5 (mg/kg), respectively. The majority of potentially toxic metals were elevated compared to background concentrations, with the exception of Co and V. Mean enrichment factors decreased in the following order: Zn > Pb > Cu > Cd > As > Cr > Ni > Co > V, with a high enrichment value for Zn and Pb. Cadmium had a moderate potential ecological risk compared to other toxic metals. Correlation coefficients and principal component analysis (PCA) identified three principal groups as sources of toxic metals in Abadan street dust. Metals such as Cr, Ni, Co and V were identified to originate from natural parent materials, Zn, Cu and Pb from anthropogenic sources emitted from combustion/vehicular materials and Cd and As from other anthropogenic sources such as the Abadan refinery and petrochemical complex. The sum of 13 PAHs ranged from 400 to 11766 (µg/kg), with a mean of 2436 (µg/kg). Approximately 68%–79% of ∑PAHs comprised of higher molecular weight PAHs (HMWPAHs 4–6 rings), revealing a combustion origin. Principal component analysis (PCA) for PAHs in Abadan street dust originated mainly from vehicular, petroleum emission and fossil fuel. The toxic equivalency quantities (TEQs) of PAHs in the street dust ranges from 22 to

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951 (μg/kg). High coefficients of correlation (r^2= 0.98 to 0.83, p<0.01) among Pyrene, Benzo (a) anthracene, Chrysene, Benzo (a) pyrene, Benzo (b) fluoranthene, Benzo (k) fluoranthene, and TEQ of street dust showed that these PAHs were principal contributors to TEQ. Ecosystem risk calculations for PAHs were high since the value of $RQ_{\sum PAHs_{(MPCs)}}$ was ≥ 1 and $RQ_{\sum PAHs_{(NCs)}}$ were ≥ 800. The mean values for $RQ_{\sum PAHs_{(MPCs)}}$ and $RQ_{\sum PAHs_{(NCs)}}$ were 8 and 1151, which indicated a relatively high ecosystem risk of PAHs in street dust in Abadan. The total cancer risk was 8.64×10^{-3} for adults and 8.95×10^{-4} for children, suggesting a high potential cancer risk through both dust dermal contact and ingestion pathways.

Keywords: Street dust, Toxic metal, PAHs, Pollution assessment, Health risk assessment, Abadan

1. Introduction

Emissions from activities relating to rapid urbanisation, industrial plants and vehicular traffic in megacities provide challenges for urban and residential environments. Urban street dust particles are one of the indices for monitoring environmental urban contaminants (Brown and Peake, 2006; Nazarpour et al., 2017b; Babaei et al., 2018), such as toxic metals and polycyclic aromatic hydrocarbons (PAHs) that present potential harm to community health (Tang et al., 2013, Saeedi et al., 2012; Soltani et al., 2015). Dermal and hand to mouth contact can result in direct exposure to such contaminated dust particles, especially in the case of children in school grounds, parks and city side roads (Ghanavati et al., 2018). The source of toxic metals in street dust can be petroleum, industrial sources, vehicle emissions and geogenic/geochemical routes, such as weathering of parent materials, construction, roadside and atmospheric dust accumulation (Gunawardana et al., 2012, Nazarpour et al., 2017b). Toxic elements are not decomposable and reside in the soil/dust environment for a long time or can be re-suspended into the atmosphere, presenting a potential hazard to human health (Cook et al., 2005, Zarasvandi et al., 2011; Ghanavati., 2018). PAHs comprise a cluster of carbon-based/organic complexes with various aromatic rings and do not contain a heteroatom or carry substituents (Kamal et al., 2015). They can present a major and permanent hazard to environmental security and human health, and for these reasons they are of major concern in recent studies (Dong and Lee, 2009; Nazarpour et al., 2017b). The sources of PAHs include natural processes (wood fires and volcanic eruptions) and anthropogenic activities (e.g., straw and fuel wood combustion, waste incineration, vehicle exhaust, combustion of coal, spillage of petroleum, diesel oil and pyrolysis of organic matter) (Abdel-Shafy and Mansour, 2016; Li et al., 2006). Sixteen parent PAHs are classified as significant contaminants and persistent toxic substances (PTS) by the United States Environmental Protection Agency (US EPA) and United Nations Environment Program (UNEP) because of their durability, potential toxicity, exhibit mutagenic and carcinogenic properties and are widely distributed in the natural environment in matrices such as dust (UNEP, 2002; Kamal et al., 2015). In order to formulate sufficient measures to decrease environmental risk of exposure to PAHs, it is essential to quantitatively assess contributions from several sources of PAHs, such as pyrogenic or petrogenic consistent with the major PAH sources to the environment

(Wang et al., 2009). Commonly, PAHs resulting from petrogenic sources contain 3–4 ring structures and have petroleum products or crude oil as their main sources. Whereas, PAHs produced by pyrogenic sources contain 4–6 ring constructions, formed when organic matters are exposed to high temperatures under low oxygen or no oxygen circumstances. The distillation of coal into coke and coal tar, or the thermic cracking of petroleum sediment into lighter hydrocarbons are pyrolytic processes that occur in urban areas (Liu et al., 2007). PAHs present the main carcinogen and cancer risk to metropolitan environments when present in urban dust, particularly when there are several anthropogenic sources. PAHs with a high molecular weight have a greater carcinogenic toxicity compared to low molecular weight PAHs (Colombo et al., 2006; Szabová et al., 2008). Multivariate analysis including correlation coefficients and principal component analysis (PCA) have been utilised to identify the source of toxic elements and PAHs (Long et al., 2013). Contamination of toxic metals and PAHs in urban dust have been considered extensively in industrialised countries (Lee et al., 2011). Abadan is one of the most significant cities in Iran, owing to its existence as an industrial hub, with a large influence from the Petrochemical Industries and Oil Refining Company. However, there is no published data about toxic elements and PAH composition of street dust from Abadan and its potential health risk. Therefore, the main objectives of this study were: (1) to determine the concentration and source of toxic metals and PAHs in street dust samples collected from Abadan city and to evaluate their pollution level; (2) to define their specifications and possible combustion sources (3) to evaluate the distribution of pyrogenic and petrogenic PAHs in dust street samples; and (4) to evaluate potential ecosystem risk and the cancer risk possibility of PAH exposure through dermal contact, ingestion and inhalation.

2. Materials and methods

2.1. Study Area

Abadan (30°20'40" N and 48°17'20"), as one of industrial cities of Khuzestan province with about 340000 inhabitants in 2014 is located in south west Iran on the border with Iraq, Kuwait and Saudi Arabia, whose deserts are the main source of dust storm occurrences over Iran.

2.2. Sample collection and preparation

In August 2016, thirty (30) street dust samples were collected from various locations in the city of Abadan (Fig.1). Samples were selected in the driest month of the year to avoid rain-washing away street dusts. Samples were collected nearby to: a refinery, petrochemical industries, an airport, and commercial and busy traffic areas of the city. Approximately 500 g of a composite street dust sample was collected by brushing an area of 2×2 m^2 from the sidewalk. The samples were air-dried and passed through a 63μm mesh. Geographical coordination of sample locations was recorded at each sample point with a handheld GPS device.

2.3. Extraction and analysis of toxic metals

Toxic metals (Ni, As, V, Cr, Pb, Cu, Co, Zn and Cd) were analyzed by Inductively Coupled Plasma-Mass Spectrometry (ICP-MS; Perkin-Elmer Elan 9000). In order to measure toxic metals, a wet digestion process was adapted from Liang and Grégoire (Liang and Grégoire, 2000). Quality assessment and quality control contained analytical duplicate samples, a reagent blank, standard reference materials GSS-17 and GSS-25 (multi-element soil standard OREAS45EA and OREAS24P) and procedural blanks. Precision, specified by duplicate measurements, was < 5% relative standard deviation (RSD) and accuracy for the reference material, was between 95% and 110% for all metals studied.

2.4. Analysis of PAHs

Thirteen PAHs specified by the US EPA as precedence contaminants were analyzed, including Phenanthrene (Phe), Anthracene (Ant), Pyrene (Pyr), Benzo (a) anthracene (BaA), Chrysene (Chr), Benzo (b) fluoranthene (B(b)F), Benzo (k) fluoranthene (B(k)f), Benzo (a) pyrene (BaP), Indeno (1,2,3-cd) pyrene (IcdP), Dibenzon (a,h) anthracene (DBA), Benzo (ghi) pyrylene (BghiP).The PAHs in street dust samples were measured using gas chromatography/mass spectrometry system (GC/MS) in Jundishapur University of Medical Sciences Comprehensive Laboratory (JUMSCL) pursuant to EPA SW-846 procedure 3550C (US EPA, 2007) and EPA 8270D (US EPA, 1998). A sample of about 10 g was extracted using a solvent combination of dichloromethane and acetone (1:1 by volume, insecticide grade, Merck). The samples were exposed to end-over-end tumbling (30 min, about 1 cycle/s) overnight. The extract was filtered via sodium sulphate followed by concentration by evaporation using

a Turbovap (Caliper Life Sciences) at 40 °C. The extracts were spiked with the USEPA8270 internal standard mix (Supelco, Sigma Aldrich, USA) and examined by gas chromatography mass spectrometry (Agilent 5975, USA) with capillary column DB5ms (30 ms, 0.25 mm, 0.25 μm film, J&W, USA). The GC/MS was operated in Selected Ion Mode with helium as the carrier gas (1 ml/min, BOC gases). The oven temperature was held at 40°C for 1 min, increased to 310°C at 18°C/min and then held for 7 minutes at this temperature. Laboratory procedural blanks were analysed with each batch of samples, and limits of quantification were defined as three times greater than the blank levels. Duplicate samples, laboratory control samples (LCS) and matrix spikes were analysed with each batch, with all analyses within ±20% relative percent difference (RPD) for duplicates (at the 10 for limits of Report (LOR's) level), and accuracies of 95–117% for LCS.

2.5 Quality control

All 30 street dust samples were examined in duplicate. Filter blanks were lower than 10% of the sample quantity. The relative standard deviation was lower than 10% for all measured PAHs. Standard reference material of street dust (SRM 2585) from the National Institute of Standards and Technology (NIST) USA was applied for calibration and analytical control. PAHs were identified on the basis of retention times relative to five deuterated internal standards (d8-Nap, d10-Ace, d10-Phe, d10-Chr, and d12-Per). Recoveries of SRM and internal standards ranged from 75% (Pyr) to 133% (Nap).

2.6. Pollution assessment

2.6.1. Enrichment factor

An enrichment factor (EF) method was applied to determine the degree of toxic metal contamination (Yuen et al., 2012) and define their sources, as either parent materials or human/anthropogenic sources (Liu et al., 2014). Enrichment factor (EF) was estimated as follows:

$$EF = \frac{\left(\frac{\mathrm{Ci}}{\mathrm{Cref}}\right)\mathrm{sample}}{\left(\frac{\mathrm{Ci}}{\mathrm{Cref}}\right)\mathrm{background}} \qquad (1)$$

155 The $\left(\frac{Ci}{Cref}\right)$ is the ratio of value of toxic metal (mg/kg) to the value of target toxic metal in the selected
156 and background sample. In this paper, Al was used as the normaliser and contents of metals in the crust
157 were derived as background (Hsu et al., 2016; Yongming et al., 2006). Consistent with (Zhang and Liu,
158 2002), EF values 0.05 ≤EF≤ 1.5 show that the toxic metal originated completely from crustal materials
159 or a natural source, although values greater than 1.5 would show anthropogenic source of toxic metals.
160 Degree of pollution can be categorised in five classes (Namaghi et al., 2011): (1) EF <2 (states
161 deficiency to minimal enrichment); (2) 2≤ EF ≤5 (moderate enrichment); (3) 5≤ EF ≤20 (significant
162 enrichment); (4) 20≤ EF ≤40 (very high enrichment) and (5) EF>40 (extremely high enrichment).

163 **2.6.2. Potential ecological risk (PER)**

164 The Potential Ecological Risk (PER) of toxic metals was originally presented by Hakanson (1980) to
165 evaluate the degree of toxic metal contamination in street dust (Qiu, 2010; Sun et al., 2010). The PER
166 was estimated as follows:

167 $$C_f = {}^{C_S}/_{C_n} \qquad (2)$$

168 $$E_r = T_r \times C_f \qquad (3)$$

169 $$RI = \sum_{i}^{n} E_r \qquad (4)$$

170 Where, RI is the sum of potential ecological risk for multiple toxic metals; E_r is the potential ecological
171 risk of each toxic metal; T_r is the toxicity response factor of each toxic metal (Cd, Cu, Pb, Cr, As, Zn,V
172 and Ni are 30, 5, 5, 2, 10, 1, 2 and 5) (Wang et al., 2011b); C_f is the contamination factor; C_S is the
173 concentration value of each toxic metal in the street dust and C_n is the corresponding background value
174 for toxic metals. Potential ecological risk values can be classified in the following order: Er_i<40, low
175 potential ecological risk; 40 ≤ Er_i<80, moderate potential ecological risk; 80 < Er_i≤160, considerable
176 potential ecological risk; 160 < Er_i≤320, high potential ecological risk; and Er_i > 320, very high
177 ecological risk (Wu et al., 2015; Zhao et al., 2014). Risk index values can be classified as follows: RI

178 < 150, low ecological risk; 150 ≤ RI < 300, moderate ecological risk; 300 ≤ RI < 600, considerable
179 ecological risk; and RI > 600, very high ecological risk.

180 **2.6.3. Potential ecosystem risk**

181 In order to estimate risk posed by certain PAHs in the Abadan, PAHs species values detected in street
182 dust were compared against their corresponding quality values in the present study. The risk quotient
183 (RQ) approach was used to estimate the potential ecosystem risk of PAHs in street dust (Kalf et al.,
184 1997). The risk quotient (RQ) values were defined in Eq. 5.

185 $$\mathrm{RQ} = \frac{\mathrm{C_{PAHs}}}{\mathrm{C_{QV}}} \qquad (5)$$

186 Where C_{PAHs} was concentration of certain PAH in street dust samples and C_{QV} was the corresponding
187 quality values of certain PAHs in the medium. The negligible concentration of (NCs) and the maximum
188 permissible concentrations (MPCs) were used as the quality values in the medium (Kalf et al., 1997).
189 Therefore, $\mathrm{RQ_{(NC_S)}}$ and $\mathrm{RQ_{(MPC_S)}}$ were calculated in Eq. 6 and 7.

190 $$\mathrm{RQ_{NC_S}} = \frac{\mathrm{C_{PAHs}}}{\mathrm{C_{QV(NC_S)}}} \qquad (6)$$

191 $$\mathrm{RQ_{MPC_S}} = \frac{\mathrm{C_{PAHs}}}{\mathrm{C_{QV(MPC_S)}}} \qquad (7)$$

192 Where C_{PAHs} was concentration of certain PAH in street dust samples. $\mathrm{RQ_{(NC_S)}}$ and $\mathrm{RQ_{(MPC_S)}}$ were the
193 quality values of the NCs and MPCs of PAHs in street dust. The RQ could only be used to evaluate the
194 ecosystem risk of 8 individual PAHs. In order to evaluate the ecosystem risk of other 5 individual PAHs
195 [acenaphthylene (Acy), Fluorene (Fl), pyrene (Pyr), benzo (b) fluoranthene (B(b)F) and dibenzo (a,h)
196 anthracene (DBA), the toxic equivalency factors (TEFs)] (Nisbet and Lagoy, 1992) were used to infer
197 that the NCs and MPCs of Acy, Fl, and Pyr (TEFs = 0.001) were similar to anthracene (Ant) (TEFs =
198 0.001), B(b)F(TEFs = 0.1) was similar to benzo(a)anthracene (BaA) (TEFs = 0.1) and dibenzo
199 (a,h)anthracene (DBA) (TEFs = 1) was similar to benzo(a)pyrene (BaP) (TEFs = 1). $RQ_{\sum PAHs}$,
200 $RQ_{\sum PAHs_{(NCs)}}$ and $RQ_{\sum PAHs_{(MPCs)}}$ were defined in Eq. 8-10 (Cao et al., 2010):

201 $$RQ_{\sum PAHs} = \sum_{i=1}^{13} RQ_i \quad (RQ_i \geq 1) \qquad (8)$$

202 $$RQ_{\sum PAHs_{(NCs)}} = \sum_{i=1}^{13} RQ_{(NCs)} \quad (RQ_{(NCs)} \geq 1) \qquad (9)$$

203 $$RQ_{\sum PAHs_{(MPCs)}} = \sum_{i=1}^{13} RQ_{(MPCs)} \quad (RQ_{(MPCs)} \geq 1) \qquad (10)$$

204 In principle, $RQ_{(NCs)}$ <1 indicated that the single PAHs are likely of negligible concern, while RQ
205 $_{(MPCs)}$>1 would indicate that contamination of the single PAHs was of more concern and some control
206 measures and remedial actions must be undertaken at once. Where $RQ_{(NCs)}$ >1 and $RQ_{(MPCs)}$ <1,
207 contamination of single PAHs might be considered of medium concern and some control measures or
208 remedial actions are required (Table 1).

209 **2.6.4. Cancer risk assessment**

210 PAH toxicity evaluations in the street dust samples were assessed applying relative toxicity values for
211 individual PAH species. The PAH toxicities of street dust samples were calculated according to the
212 collection of toxicity equivalency factors (TEFs) for PAHs (Nisbet and LaGoy, 1992). For example,
213 BaP (a potent carcinogen in the PAH group) was identified as the reference chemical and assigned a
214 value of 1, whilst other PAHs have their specific TEF values according to their carcinogenic level
215 relative to BaP (Fang et al., 2004). The carcinogenic potency of total PAHs is calculated using the toxic
216 benzo[a] pyrene equivalent (BaPeq) contents of each PAH (Lee and Dong, 2010). The toxic equivalent
217 quotient (TEQ) of each street dust sample was computed by collecting the products of each individual
218 PAH content. TEQs were calculated in Eq. 11 and 12.

219 $$\mathrm{BaPeq_i} = \mathrm{PAH_i} \times \mathrm{TEF_i} \qquad (11)$$

220 $$\mathrm{TEQ} = \sum_{\mathrm{i}}^{\mathrm{n}} (\mathrm{PAH_i} \times \mathrm{TEF_i}) \qquad (12)$$

221 Where PAH_i is the content of individual PAHs, and TEF_i is the corresponding toxic equivalency factor
222 and TEQ is the toxic equivalent of the compound.

Humans are exposed to street dust throughout three major exposure routes including, dermal contact, ingestion and inhalation. The incremental lifetime cancer risk (ILCR) model was used to quantitatively calculate the exposure risk for environmental PAHs according to the U.S. EPA standard models (Chen and Liao, 2006; Peng et al., 2011). ILCR in terms of ingestion, dermal contact, and inhalation were calculated in Equations 13–15.

$$LCRs_{Ingestion} = \frac{CS \times \left(CSF_{Ingestion} \times \sqrt[3]{\left(\frac{BW}{70}\right)}\right) \times IR_{Ingestion} \times EF \times ED}{BW \times AT \times 10^{6}} \quad (13)$$

$$LCRs_{Dermal} = \frac{CS \times \left(CSF_{Dermal} \times \sqrt[3]{\left(\frac{BW}{70}\right)}\right) \times SA \times AF \times ABS \times EF \times ED}{BW \times AT \times 10^{6}} \quad (14)$$

$$LCRs_{Inhalation} = \frac{CS \times \left(CSF_{Inhalation} \times \sqrt[3]{\left(\frac{BW}{70}\right)}\right) \times IR_{Inhalation} \times EF \times ED}{BW \times AT \times PEF} \quad (15)$$

Where CS is the sum of converted PAH levels according to toxic equivalents of BaP applying the toxic equivalency factor (TEF) (Fang et al., 2004; Nisbet and LaGoy, 1992). CSF is the carcinogenic slope factor (mg/kg/day),;BW is body weight assumed to be 15 kg for children and 61.5 kg for adults (Shi et al., 2011); AT is the mean life span being 70 × 365 = 2550 (days) (Ferreira-Baptista and De Miguel, 2005); EF and ED are exposure frequency and duration, respectively. The EF value is 180 (day/year) for children and adults (Ferreira-Baptista and De Miguel, 2005); ED value is 24 years for adults and 6 years for children (USEPA, 2001); $IR_{Ingestion}$ is the dust intake rate at 200 (mg/day) for children and 100 (mg/day) for adults (U.S. EPA 2001), $IR_{Inhalation}$ is the inhalation rate at 10 (m^3/day) for children and 20 (m^3/day) for adults (USEPA, 2001), SA is the dermal surface exposure; SA value is 2800 (cm^2/day) for children and 5700 (cm^2/day) for adults, respectively. AF is the dermal adherence factor, AF which is 0.2 (mg/cm) for children and 0.07 (mg/cm) for adults (USEPA 2001); ABS is the dermal adsorption fraction (unit less) being 0.13 (U.S. EPA 2001), and PEF is the particle emission factor defined as 1.36×10^9 (m^3/kg) (USEPA 2001). The designation of CSF, according to the cancer-causing ability of BaP; $CSF_{Ingestion}$, CSF_{Dermal} and $CSF_{Inhalation}$ of BaP, are 7.3, 25 and 3.85 (mg/kg/day), respectively (Peng et al., 2011). All the parameters used in these models for children (1 to 6 years old) and adults (7 to 31

246 years old) were according to the Risk Assessment Guidance of U.S. EPA and related publications
247 (Wang et al., 2011a).

248 **2.7. Evaluation of PAH sources**

249 The ratios of designated isomeric PAHs have been extensively applied to recognise PAH sources
250 (Ravindra et al., 2008). In this paper, the attribution of pyrogenic or petrogenic sources to the PAHs
251 found in the street dust fractions were based on two criteria: a) the ratio of ΣCOMB to ΣPAHs
252 (ΣCOMB/ΣPAHs) can be applied to recognise the source of PAHs, where ΣCOMB is the sum of the
253 main combustion specific compounds. These include Fl, BaA, Pyr, B(k)F, Chry, B(b)F, IcdP, BaP and
254 BghiP and ΣPAH as the sum of PAH concentrations (Kamal et al., 2015). Spearman's correlation
255 analysis and principal component analysis (PCA) were carried out using the statistical software package
256 SPSS version 22.0 for windows. The Spearman's correlation coefficient was applied to measure the
257 strength of relationships between pairs of PAHs within samples. A significant and positive correlation
258 between PAHs often shows that the PAHs are derived from the same sources (Saeedi et al., 2012; Wang
259 et al., 2011a).

260 **3. Results and discussion**

261 **3.1. Toxic metal concentrations**

262 Statistical concentration data for toxic metals in street dust from Abadan is compared to
263 background/upper continental crust (UCC) values in Table 2. Mean concentrations for Pb, Zn, Cu, Cr,
264 Cd, Ni, V, As and Co in road dust were: 59, 288, 113, 50, 0.5, 57, 36, 7 and 8 mg/kg, respectively. With
265 the exception of Co and V, all metals were higher than their corresponding UCC, indicating possible
266 anthropogenic input of metals from vehicular traffic, industrial plants and construction activities. A
267 Kolmogorov–Smirnov test indicated that all toxic metal concentrations were non-normally distributed
268 (significant level > 0.05) except for Zn, Cr and V (significant level < 0.05). Skewness values for all
269 toxic metals were positive, with only Cr, Ni and Co displaying normal distribution. The skewness for
270 Cu was 4.83 (highest value among all of the toxic metals), which shows the existence of extremely

polluted spots. Based on the coefficient of variation (CV), the toxic elements analysed can be categorised into two groups: Pb, Zn, Cu, Cd and As with a CV greater than 0.4; and Co, Cr, Ni and V with a CV lower than 0.4. Elements dominated by a natural source exhibit a low CV, while those affected by anthropogenic sources are expected to have a high CV (Wang and Lu, 2011; Yuan et al., 2014). There is no standardised protocol for the sampling and geochemical analysis of urban street dust, and specifically such values do not exist in the study area of Abadan. The mean concentrations of pollutants in street dust are generally compared to other urban environments rather than a defined threshold value. Mean concentrations of metals in Abadan street dusts were compared with other cities in the world in Table 3. The mean concentrations of Pb and Zn in Abadan street dusts in this study are higher than Ottawa and Beijing, but lower than Tehran, Shiraz, Xian, Hong Kong, Shanghai, Madrid and Amman. The mean concentration of Cu is lower than Tehran, Shiraz, Shanghai, Madrid and Amman, and higher than Xian, Hong Kong, Ottawa and Beijing. Generally, every urban area has its own typical signature for toxic metal combinations resulting from variations in anthropogenic and natural geogenic sources.

3.2. Toxic metal contamination in street dust

Upper crust (UCC) values were applied as background concentrations for toxic metal enrichment factors (EF) calculations, as reported in previous studies (Kartal et al., 2006; Reimann and de Caritat, 2005). In this regard, Al was taken as the reference element. The range of EF values for Pb, Zn, Cu, Cr, Cd, Ni, V, As and Co were 7–122, 20–157, 7–235, 8–14, 12–33, 17–24, 4–7, 6–43 and 5–6, respectively (Fig. 2). The average EF values decreased as follows: Zn (68) > Pb (29) > Cu (28) > Cd (19) > As (11) > Cr (9) > Ni (8.7) > Co (5)> V (4). The mean EF values for Pb, Zn, Cu, Cd, and As in Abadan street dust samples were > 10, suggesting anthropogenic sources for these metals in the study area, whereas Ni, Cr, V and Co can be considered to originate primarily from natural geogenic sources, such as wind-blown soil minerals. Zinc exhibited the highest EF mean (> 40), suggesting extremely high enrichment. These results demonstrate that the toxic metals reflect anthropogenic sources emitted from industrial plants and vehicular materials, such as brakes and tyres (Kong et al., 2011).

297 In this study, the potential ecological risk (Er) of toxic elements was evaluated by applying the
298 Hakanson technique (Hakanson, 1980). According to the trend of *Er* in dust samples: Cd (78) >Pb (20)
299 >Cu (19) >As (15) > Ni (14) >Zn (9)>Cr (3) > V (1). Mean Er values for Cu, Pb, As, Ni, Zn, Cr and V
300 were < 40, which suggested a low risk for these metals, thus indicating a low potential *Er*. In contrast,
301 Cd has a moderate potential Er compared to other toxic metals (Table 4). The potential ecological risk
302 indices (*RI*) values were evaluated to assess the risk of multiple-metals associated with street dusts. In
303 general, *RI* describes the sensitivity of the environment to potentially toxic metals providing an overall
304 indication of pollution. The *RI* shows that values for 57 % of all samples were at a low risk, 40% and
305 3% were at moderate and considerable risk, respectively (Table 5).

306 **3.3. Toxic metal source identification**

307 Inter-element relationships between elements provides useful information about the sources and
308 pathways of toxic metals (Huang et al., 2009; Lu et al., 2010). Spearman's correlation coefficients of
309 toxic metals in Abadan street dusts are presented in Table 6. One group of toxic metals, Ni, Cr, Co and
310 V, show a very significant correlation with each other at $p < 0.01$: Co-V (0.90), Co-Ni (0.85), Ni-V
311 (0.83), Co-Cr (0.79), Cr-V (0.79) and Ni-Cr (0.70), suggestinga common source. A significant
312 correlation ($p < 0.01$) was also found between a second group of toxic metals, Pb, Zn, Cu, Cd and As
313 including Pb-Cu (0.86), Cu-Zn (0.84), Pb-Zn (0.68), Pb-Cd (0.56), Cu-Cd (0.51), Cd-As (0.51), Pb-
314 As(0.47) and Cu-As(0.47), which indicated a different source.

315 In this study, principal component analysis (PCA) was employed to further understand the relationship
316 between metals in urban dust and their possible sources for each factor. Bartlett's test of sphericity was
317 significant at $p < 0.001$ and 0.569 for the Kaiser–Meyer– Olkin Index, which confirmed that toxic
318 metals associated with urban dust was appropriate for (PCA). The result of principle component
319 analysis (PCA) with varimax rotation was used for toxic metals concentrations. Three Principal
320 Components (PC) accounting for 83% of the total variance is presented in Table 7. The first principal
321 component (PC1) explains 42% of the total variance is dominated by Cr, Ni, Co and V. As well as, the
322 average values of EF for Cr, Ni, Co and V are essentially lower than for the other toxic elements. As a

result, these toxic metals likely originated from natural sources. The second component (PC2) accounting for 27% of total variance is characterised by Zn, Pb and Cu. In the loading plot (Fig. 3). A third (PC3) explains 14% of total variance, and is dominated primarily by As and Cd. However, the high values of EF for As and Cd proposes that anthropogenic sources are the main sources of these toxic metals.

3.4. Spatial distribution patterns of toxic metals

The spatial distribution of toxic metals concentrations is presented in Figure 4. The spatial distribution pattern of toxic metals concentrations can aid in identifying areas above UCC and to determine the sources of contamination. Considering the activity of the two large refineries and petrochemical complexes and the arrival and accumulation of a large number of toxic metal compounds, it was expected that the highest concentrations of toxic metals in street dust around the industrial zones. However, spatial distribution of toxic metals and its adaptation to Abadan urban zoning showed that relatively high concentrations of toxic metals occurred instead in dense residential areas with high traffic volumes and urban shopping centres. According to the weather data, annual rainfall in Abadan is about 157 (mm/year) and is mostly dry and frequented by prevailing winds from the northwest through all seasons, but with more intensity in the summer. The spatial distribution of metals in street dust suggested that the prevailing wind has a significant influence on the distribution of toxic metals across Abadan. As a result, toxic metals are transferred from the areas around the refinery and petrochemicals to urban and residential areas.

3.5. PAH concentrations

Descriptive statistics for 13 PAHs in Abadan street dust samples are summarised in Table 8. Total PAH content (Σ13PAHs) in Abadan street dust were in a range of 400–11766 (μg/kg) with a mean of 2436 (μg/kg), showing that PAHs are inclined to accumulate in street dust and that street dust can be applied as an index of urban environmental contamination. The mean value for PAH species in Abadan street dust decreased as follows: Pyr (753) > Phe (382) > Chr (278) > Fl (167) > Icdp (155) > B(b)F(153) (μg/kg). Therefore, Pyr, Phe, Chr, Fl, Icdp, B(b)F are the major components. However, the lowest

349 measurable species in street dust are BaP and DBA with mean concentrations of 61 and 46 (μg/kg).
350 Combustion PAHs (ComPAHs, the sum of Pyr, Phe, BaA, Fl, Chr, BaP and B(b + k)F after (Xiao et al.
351 2014) make up a considerable proportion of the total content of PAHs in measured samples, a range of
352 0.73 to 0.80 (μg/kg). The ratio of carcinogenic PAHs— B(k)F, B(b)F, BaA, BaP and Chr (Rajput and
353 Lakhani, 2009) to total content of PAHs (CanPAHs / ∑13PAHs) range from 0.20 and 0.40. The ratio
354 of non-carcinogenic PAHs to total content of PAHs (NCanPAHs / ∑13PAHs) ranging from 0.60 and
355 0.80. The street dust content of total PAHs (2436 μg/kg) in this study was greater when compared with
356 urban Tehran, Iran (330 μg/kg) (Saeedi et al., 2012), Bangkok, Thailand (1100 μg/kg)
357 (Boonyatumanond et al., 2007), Isfahan, Iran (1074 μg/kg) (Soltani et al., 2015), Hanoi, Vietnam (1500
358 μg/kg) (Tue et al., 2014). However, concentrations of ∑13PAHs in urban Abadan street dust was lower
359 compared with urban Sydney, Australia (2910 μg/kg) (Nguyen et al., 2014), Lanzhou, China (3900
360 μg/kg) (Jiang et al., 2014), Guangzhou, China (4800 μg/kg) (Wang et al., 2011a), Xian, China (10620
361 μg/kg) (Wei et al., 2015).

362 In general, PAHs in urban dust is classified with the number of aromatic rings, three-ring PAHs (Ant,
363 Phe, Fl and Acy) contributed 30 % of ∑PAHs, four-ring PAHs (BaA, Pyr and Chr) contributed 47 % of
364 ∑PAHs, Among five-ring (B(b)F,B(k)f, BaP and IcdP) to six-ring (DBA and BghiP) PAHs, contributed
365 19%, and 5 %, respectively. The average content of lower molecular weight PAHs (LMW 3 rings) is
366 737 (μg/kg), contributed 30% of ∑PAHs, although the mean content of higher molecular weight PAHs
367 (HMW 4–6 rings) was 1699 (μg/kg), contributed 70% of ∑PAHs, indicating a pyrogenic source
368 (Zakaria et al., 2002). The contribution of high molecular weight PAHs was significant and likely from
369 a common origin of vehicle emission (Hassanien and Abdel-Latif, 2008). In addition, this finding was
370 consistent with individual PAHs detected in gasoline engine soot enriched by high molecular weight
371 PAHs, indicating a prevailing effect of gasoline engine release (Kong et al., 2012).

372 **3.6. PAHs source identification**

373 Spearman's correlation coefficients among PAH species, ∑PAHs and TEQ in Abadan street dusts are
374 presented in Table 9. The majority of variables were significantly correlated ($p < 0.05$), although,

stronger correlations characterised by $r > 0.7$ occurred for HMW species ($p < 0.01$). Although, the stronger correlations characterised by r-values ($r>0.8$) were detected only among HMW species group, and concurrently correlated with ∑PAHs. LMW compounds exhibited a negligble correlation with HMW compounds, indicating various sources for these two groups. Significant correlation coefficients between HMW and TEQs in Abadan street dust ($r^2 = 0.932$, 0.433 and 0.980, $p < 0.01$) show that LMW compounds tend to have lesser toxicity than HMW compounds (Anastasopoulos et al., 2012).

In this study, principal components analysis (PCA) with varimax rotation was applied to identify the relationship among PAH compositions and possible chemical sources in urban street dust. Principal components analysis (PCA) with the varimax rotation Bartlett's test of sphericity were significant at $p < 0.001$ and results of Kaiser–Meyer– Olkin Index of 0.689 extracted three principal components (Factor 1, Factor 2 and Factor 3) and represented more than 87% of the total variance of ∑PAHs in the urban street dust (Table 10). Factor 1 (63% of the total variance) was characterised by high loadings of PAHs with four to six rings, including Chr, BaA, Pyr, B(k)F, B(b)F, BaP, IcdP, DBA, DghiP, representing vehicle-related sources (Boonyatumanond et al., 2007). BaP, Chr, and BaA are identified as markers of gasoline emissions (Larsen and Baker, 2003), although B(k)F and B(b)F are identified as markers of fossil fuels combustion (Park et al., 2002). In summary, Factor 1 was assigned to vehicular traffic contamination and petrogenic sources. Factor 2 (16% of the total variance) had high loadings of PAHs with three rings, including Acy, Fl and Phe, representing combustion sources (Steinhauer and Boehm, 1992). LMW PAHs, such as Phe, Acy, Fl, show the attendance of significant combustion products of low-temperature pyrogenic processes (Jenkins et al., 1996). Factor 3 (7% of the total variance) displayed high loading values of Ant, in which Ant represents oil combustion markers (Fig.5) (Dong and Lee, 2009).

3.7. Spatial distribution patterns of PAHs

The spatial distribution of PAHs in Abadan street dust is shown in Fig. 6. The results of the spatial distribution pattern of PAHs concentration and its adaptation to urban zoning in Abadan showed that the highest concentration of PAHs with lower molecular weight PAHs (LMW 3 rings) occurred around

these industrial centers, with high residential density and traffic volume (Abadan-Khorramshahr road). The highest concentration of PAHs with higher molecular weight PAHs (HMW 4–6 rings) was observed close to a refinery and petrochemical complex. The spatial distribution pattern showed that the highest concentrations of ΣPAHs and TEQ were around two refineries within a petrochemical complex. The results show that the most important factor in the entry and accumulation of a large amount of PAHs in Abadan street dust is due to the activity of industries (refinery and petrochemicals), combustion of oil and gas and traffic.

3.8. Potential ecosystem risk

The mean RQ $_{(MPCs)}$ values of Fl and Pyr were all higher than 1, indicating a high level of risk to the ecosystem, with severe toxicity (Table 11). The mean RQ $_{(MPCs)}$ and RQ $_{(NCs)}$ values of BghiP in street dust was less than 1, indicating a relatively low risk. The mean RQ $_{(MPCs)}$ values of other individual PAHs, such as Acy, Phe, Ant, BaA, Chr, B(b)F, B(k)F, BaP, IcdP, and DBA in street dust were all lees than 1 and RQ $_{(NCs)}$ higher than 1, indicating a moderate risk. These results indicated that 3- and 4-ring PAHs contributed to the main ecosystem risk burden in street dust. $RQ_{\sum PAHs_{(MPCs)}}$, were higher than 1 and $RQ_{\sum PAHs_{(NCs)}}$ were higher than 800. These results indicated that the ecosystem was at a high risk in Abadan. In conclusion, the results of the diagnostic ratios indicated that PAHs in Abadan originated from the activity of industries, vehicular emission; fossil fuel and petroleum emission. The results of ecosystem risk assessments indicated a high risk of Fl and Pyr in street dust. Therefore, several control measures and strict rules should be implemented by the government to decrease emissions.

3.9. Health risk assessments

Health risk assessments of the studied PAHs in the street dust samples through the three main exposure pathways including ingestion, dermal exposure, and inhalation was performed for children and adults (Peng et al., 2011). The toxic equivalency concentration (TEQ) of 13 PAHs in Abadan street dusts was assessed using the total BaPeq, each of which was computed using toxic equivalence factors (TEFs) (Tue et al., 2014). Results indicated that the dust exposure pathways of PAHs for both children and adults decrease as follows: dermal contact > ingestion >inhalation. The conclusion indicated that the

levels of cancer risk through ingestion and dermal contact range from 10^{-3} to 10^{-5} in all street dust samples, which were 10^4 to 10^5 times greater than that via an inhalation pathway (10^{-9} to 10^{-8}) (Table 12). Consequently, inhalation of suspended particles via the nose and mouth was almost negligible, when compared with the other pathways. The incremental life time cancer risk (ILCR) through ingestion are greater for children than adults due to their hand-to-mouth activity, so that polluted street dust can be more easily ingested (Jiang et al., 2014), although risks through dermal contact and inhalation are greater for adults than children. The greater incremental life time cancer risk (ILCR) for adults via dermal contact can be described by the greater values of dermal exposure area (SA) and exposure duration (ED) of adults (Wang et al., 2011a). As well as, primary expansion of organ, nervous, and immune systems could perhaps increase the carcinogens sensitivity in children (Maertens et al., 2008). Therefore, the potential health hazard for children exposed to street dust with associated PAHs was significantly higher than for adults (Wang et al., 2011a). The greater cancer risks through inhalation for adults was due to their greater exposure period and inhalation rate. It is commonly supposed that the incremental life time cancer risk (ILCR) value of $\leq 10^{-6}$ indicates a negligible risk under most regulatory programmes, a value among 10^{-6} and 10^{-4} indicated a potential risk, whereas a value > 10^{-4} is considered to be a high risk (Wang et al., 2011a). In this study, the total cancer risk is 8.64×10^{-3} for adults and 8.95×10^{-4} for children. The total cancer risks for both adults and children in the Abadan urban area indicated a high potential cancer risk through both dust dermal contact and ingestion pathways.

4. Conclusion

The concentration values, enrichment factors and ecological risk of the potentially toxic metals (Pb, Zn, Cu, Cd, Cr, As, Ni, Co and V) in Abadan street dusts were determined, with all but Co and V in street dust greater than their corresponding UCC, demonstrating potential anthropogenic sources. EF values in street dust decreased as follows: Zn> Pb > Cu > Cd > As> Cr > Ni > Co > V. Lead, Zn and Cu in street dusts demonstrated a high EF mean enrichment value, suggesting that the dusts in this study area were significantly affected by human activities (vehicular traffic, abrasion of tyres, asphalt pavement and industrial plants). Lead, Zn, Cu, Cr, As, Ni, Co and V had a low PER, whereas Cd had a moderate PER value (77.90). Based on the correlation coefficients and principal component analysis (PCA), toxic

metals in street dust were grouped into three main sources: Cr, Ni, Co and V that originate mainly from natural sources, whilst Pb, Zn and Cu originate from anthropogenic sources emitted from vehicular materials and Cd and As from wider anthropogenic activity. The total concentration of ∑13PAHs ranged from 400 to 11766 (μg/kg), with a mean of 2436 (μg/kg). Approximately 68%–79% of ∑PAHs was comprised of higher molecular weight PAHs (HMW 4–6 rings), revealing a combustion origin. Principal component analysis (PCA) suggested that the activity of industries (refinery and petrochemicals); vehicular emission; fossil fuel and petroleum emission are likely the main contributors of PAHs in Abadan street dust. Ecosystem risk assessment indicated that Acy, Phe, Ant, BaA, Chr, B(b)F, B(k)F, BaP, IcdP, and DBA widely showed middle level ecosystem risk and Fl and Pyr indicated a high level of risk to the ecosystem in street dust of Abadan. Incremental lifetime cancer risk (ILCR) estimation showed that the PAH concentrations in Abadan street dusts had a high carcinogenic risk through both dust ingestion and dermal contact pathways for children and adults.

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Figure captions

Fig. 1. Sampling site, refinery and petrochemical complex (yellow rectangle) locations in Abadan, Iran.

Fig.2. Box-plot of EF for toxic metals in Abadan street dust.

Fig. 3. Loading plots of the principal component analysis (PCA) for toxic metals in the street dust samples from Abadan.

Fig. 4. Spatial distribution of toxic metals concentration in street dust of Abadan

Fig. 5. Factor loading plots of factor analysis for PAHs in the street dust samples from Abadan.

Fig. 6. Spatial distribution of PAHs concentration in street dust of Abadan.

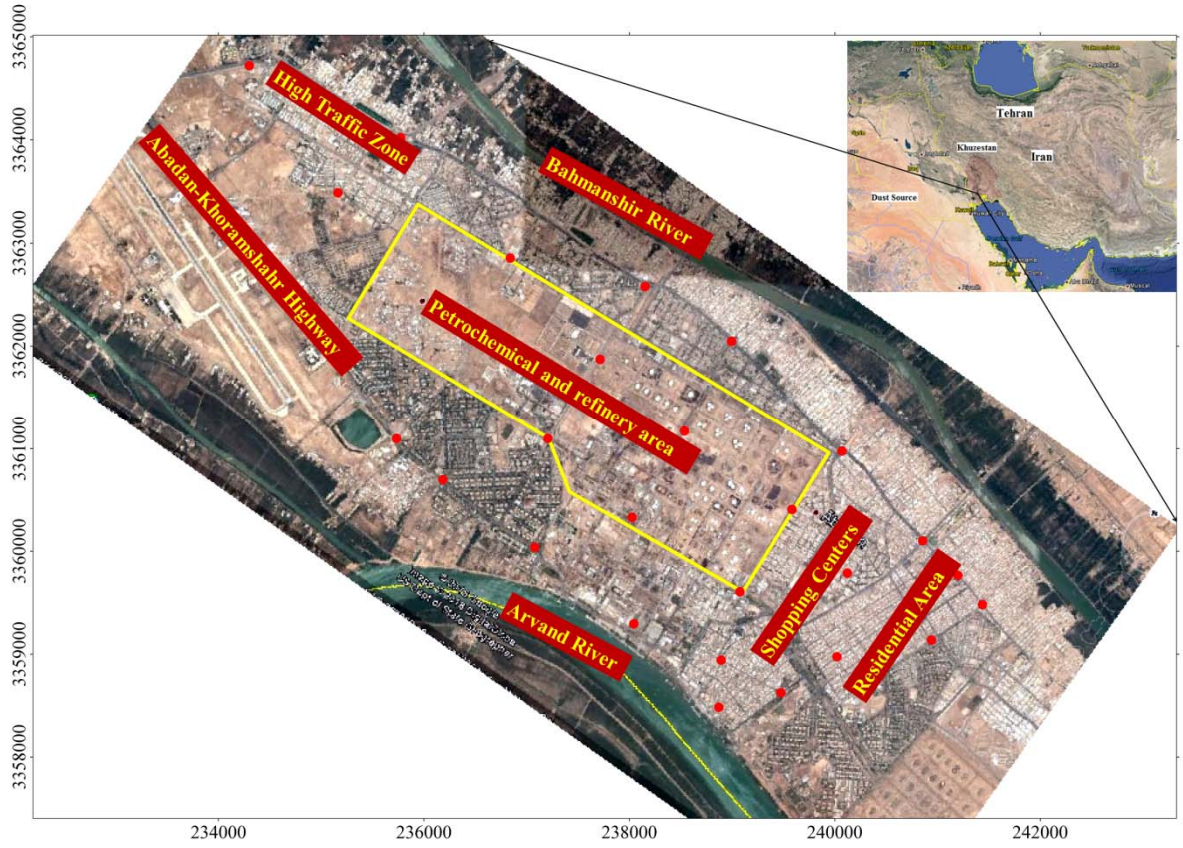
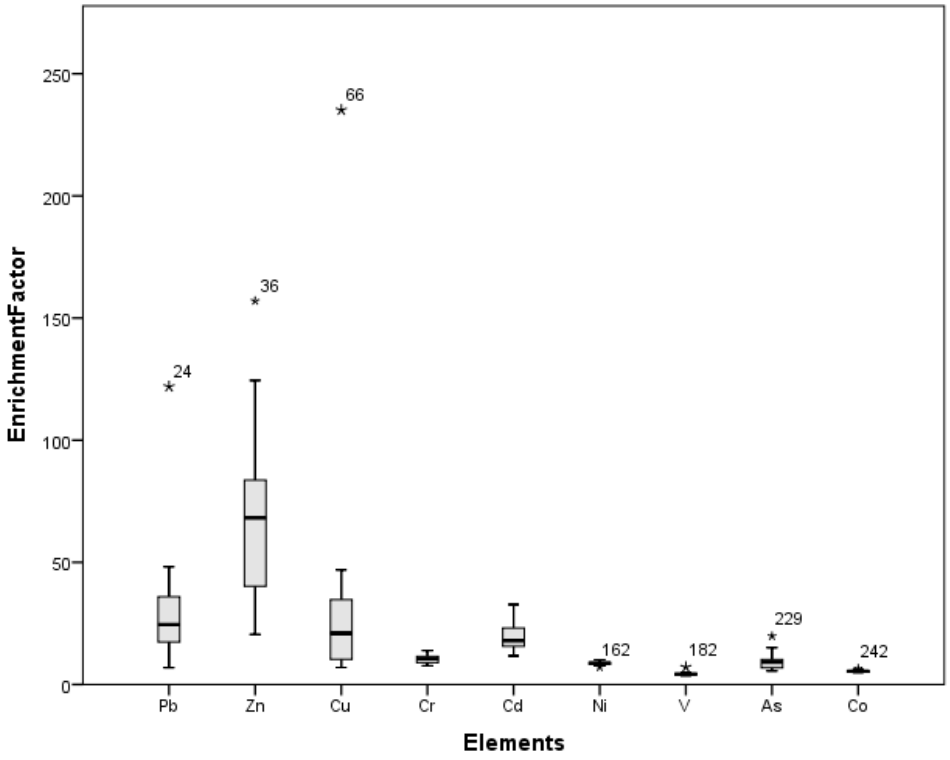

High Traffic Zone
Abadan-Khoramshahr Highway
Bahmanshir River
Petrochemical and refinery area
Arvand River
Shopping Centers
Residential Area
Tehran
Khuzestan
Iran
Dust Source
3365000
3364000
3363000
3362000
3361000
3360000
3359000
3358000
234000
236000
238000
240000
242000

Fig 1.


EnrichmentFactor
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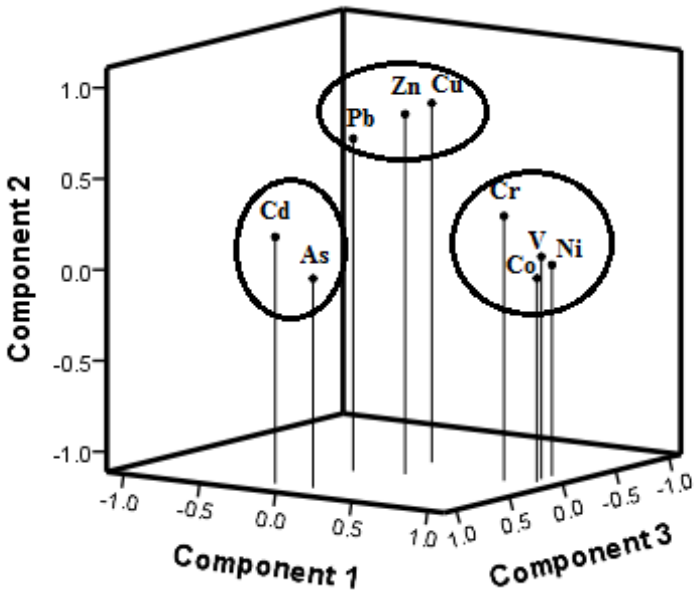
200
150
100
50
0
66
36
24
162
182
229
242
Pb
Zn
Cu
Cr
Cd
Ni
V
As
Co
Elements

Fig 2.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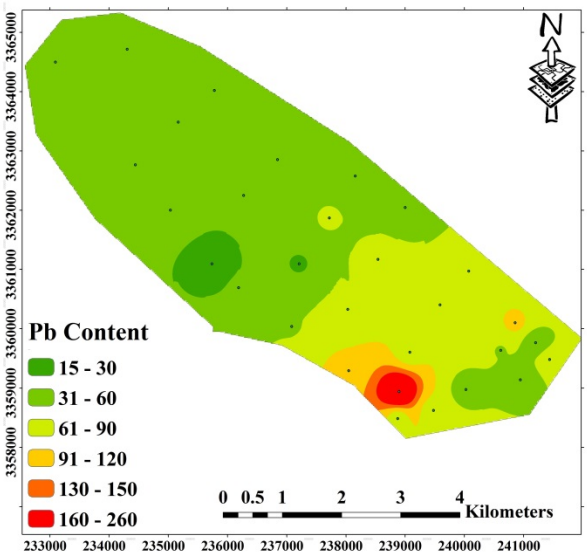
0.0
-0.5
-1.0
Zn
Cu
Pb
Cd
As
Cr
V
Ni
Co
-1.0
-0.5
0.0
0.5
1.0
Component 1
1.0
0.5
0.0
-0.5
-1.0
Component 3

Fig 3.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31 - 60
61 - 90
91 - 120
130 - 150
160 - 260
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Kilometers
N
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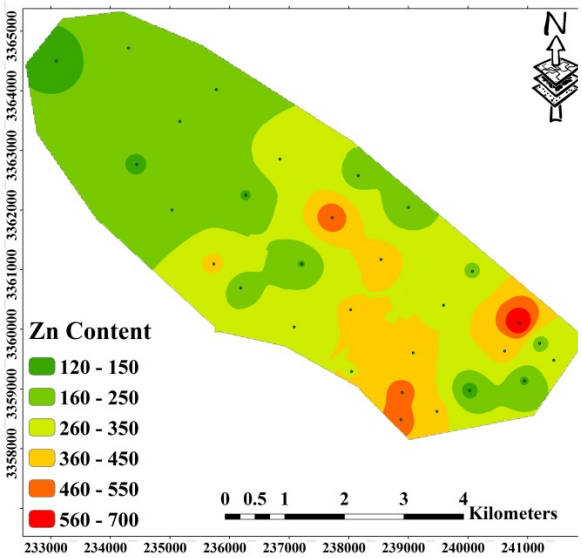
233000 234000 235000 236000 237000 238000 239000 240000 241000


Zn Content
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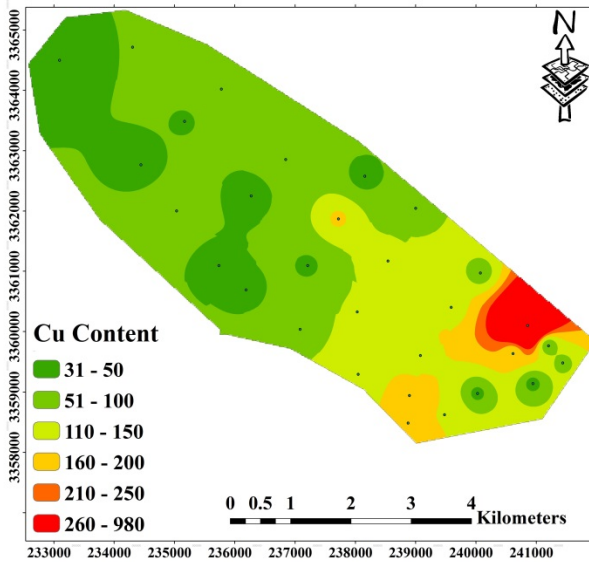
160 - 250
260 - 350
360 - 450
460 - 550
560 - 700
0 0.5 1 2 3 4
Kilometers
N
3365000 3364000 3363000 3362000 3361000 3360000 3359000 3358000
233000 234000 235000 236000 237000 238000 239000 240000 241000


Cu Content
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51 - 100
110 - 150
160 - 200
210 - 250
260 - 980
0 0.5 1 2 3 4
Kilometers
N
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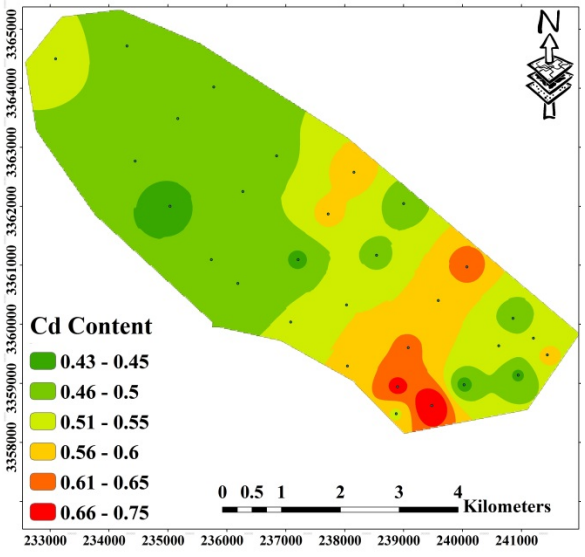
Cd Content
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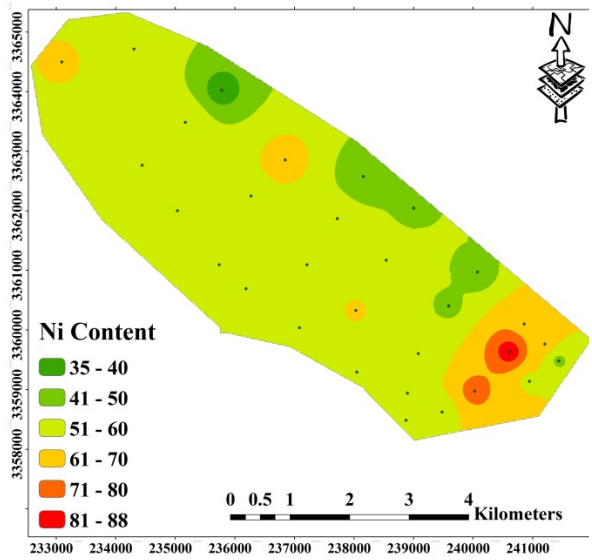
0.46 - 0.5
0.51 - 0.55
0.56 - 0.6
0.61 - 0.65
0.66 - 0.75
0 0.5 1 2 3 4
Kilometers
N
3365000 3364000 3363000 3362000 3361000 3360000 3359000 3358000
233000 234000 235000 236000 237000 238000 239000 240000 241000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51 - 60
61 - 70
71 - 80
81 - 88
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Kilometers
N
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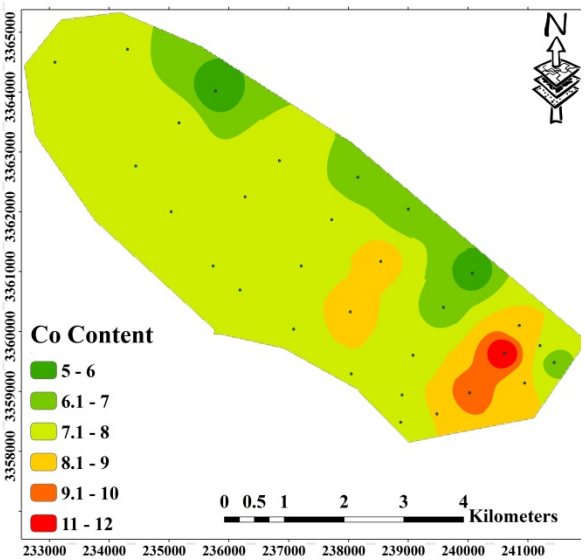
233000 234000 235000 236000 237000 238000 239000 240000 241000


Co Content
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6.1 - 7
7.1 - 8
8.1 - 9
9.1 - 10
11 - 12
0 0.5 1 2 3 4
Kilometers
N
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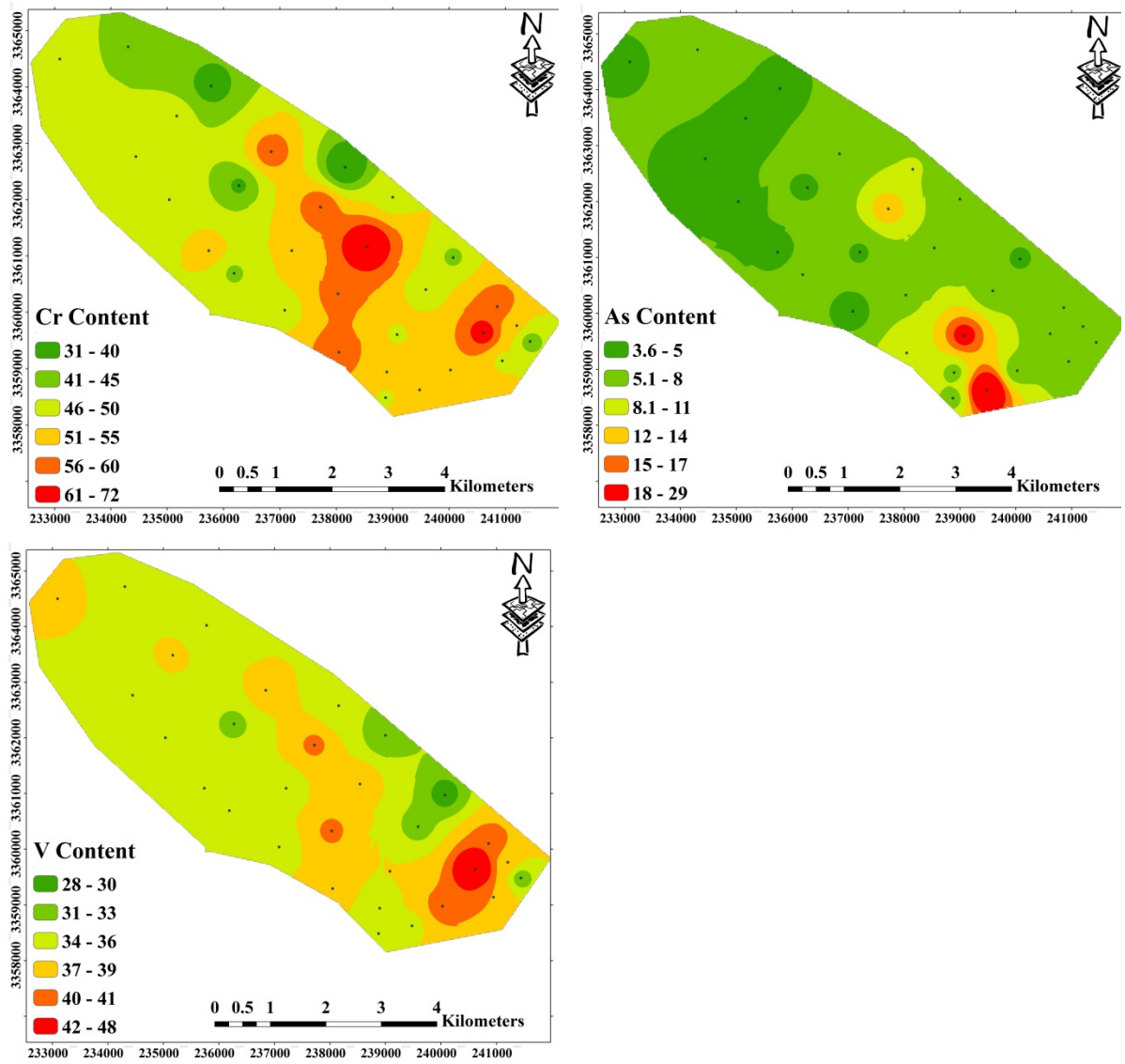
233000 234000 235000 236000 237000 238000 239000 240000 241000


Cr Content
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41 - 45
46 - 50
51 - 55
56 - 60
61 - 72
0 0.5 1 2 3 4 Kilometers
As Content
3.6 - 5
5.1 - 8
8.1 - 11
12 - 14
15 - 17
18 - 29
0 0.5 1 2 3 4 Kilometers
V Content
28 - 30
31 - 33
34 - 36
37 - 39
40 - 41
42 - 48
0 0.5 1 2 3 4 Kilometers
N
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3358000 3359000 3360000 3361000 3362000 3363000 3364000 3365000

Fig 4.

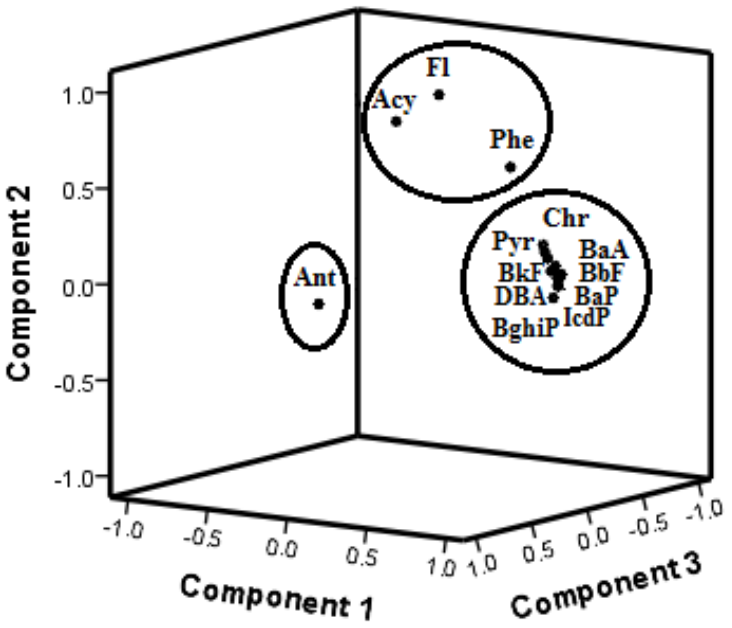
Fl
Acy
Phe
Chr
Pyr
BaA
BkF
BbF
DBA
BaP
BghiP
IcdP
Ant
Component 2
1.0
0.5
0.0
-0.5
-1.0
-1.0
-0.5
0.0
0.5
1.0
Component 1
1.0
0.5
0.0
-0.5
-1.0
Component 3

Fig 5.

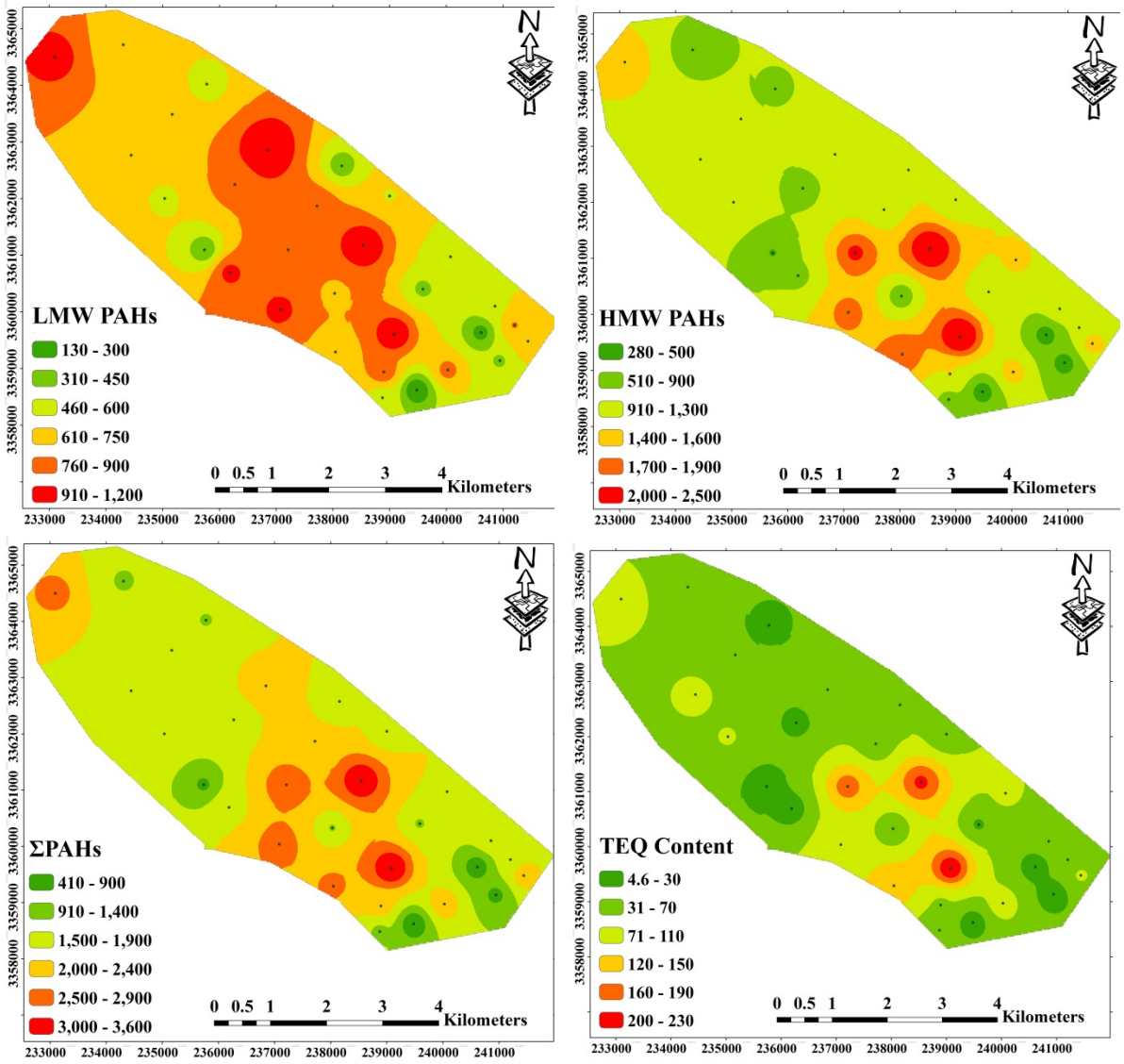
LMW PAHs
130 - 300
310 - 450
460 - 600
610 - 750
760 - 900
910 - 1,200
HMW PAHs
280 - 500
510 - 900
910 - 1,300
1,400 - 1,600
1,700 - 1,900
2,000 - 2,500
ΣPAHs
410 - 900
910 - 1,400
1,500 - 1,900
2,000 - 2,400
2,500 - 2,900
3,000 - 3,600
TEQ Content
4.6 - 30
31 - 70
71 - 110
120 - 150
160 - 190
200 - 230
N
0 0.5 1 2 3 4
Kilometers
233000 234000 235000 236000 237000 238000 239000 240000 241000
3358000 3359000 3360000 3361000 3362000 3363000 3364000 3365000

Fig. 6

Table captions

Table 1

Individual PAHs			ΣPAHs		
	$RQ_{(NCs)}$	$RQ_{(MPCs)}$		$RQ_{\Sigma PAHs\ (NCs)}$	$RQ_{\Sigma PAHs\ (MPCs)}$
Risk-free	< 1	< 1	Risk-free	< 1	< 1
			Low-risk	≥ 1 ; < 800	< 1
Moderate-risk	≥ 1	< 1	Moderate-risk1	≥ 800	< 1
			Moderate-risk2	≤ 800	≥ 1
High-risk	≥ 1	≥ 1	High-risk	≥ 800	≥ 1

Table 2

Element	Unit	Street dusts (n=30)					
		Min-Max	Mean ± SD	Skewness	CV	K-S	Upper crust content [a]
Pb	mg/kg	15 - 262	59±43	3.67	0.73	0.000	15
Zn	mg/kg	116 - 701	288 ± 139	1.01	0.48	0.065	31
Cu	mg/kg	31 - 982	113 ± 168	4.83	1.49	0.000	29
Cr	mg/kg	31 - 72	50 ± 8	0.35	0.17	0.200	35
Cd	mg/kg	0.43 - 0.75	0.52 ± 0.08	1.14	0.45	0.037	0.09
Ni	mg/kg	35 - 88	57 ± 10	0.69	0.17	0.004	47
V	mg/kg	28 - 48	36 ± 4	0.71	0.11	0.200	60
As	mg/kg	4 – 29	7 ± 5	3.31	0.72	0.000	4.8
Co	mg/kg	5 - 12	8 ± 1	0.84	0.18	0.000	10

Rollinson, (2014).

Table 3

City	Pb	Zn	Cu	Cr	Cd	As	V	Co	Ni	Reference
Tehran (Iran)	116	403	136.34	67	0.5	7	-	-	78	(Saeedi et al., 2012)
Shiraz (Iran)	257	873	225.3	34	11	-	-	-	35	(Keshavarzi et al., 2015)
Xi an (China)	231	421	94.98	167	-	11	-	-	-	(Yongming et al., 2006)
Hon Kong (China)	120	3840	110	124	-	67	37	10	29	(Yeung et al., 2003)
Beijing (china)	54	219	46	87	1.1	6	-	9	34	(Liu and Cen, 2007)
Shanghai(China)	237	753	258	264	1.0	8	-	-	66	(Shi et al., 2011)
Ottawa(Canada)	33	101	38	42	0.3	1	-	8	15	(Rasmussen et al., 2001)
Madrid (Spain)	1927	467	188	61	-	-	17	3	44	(de Miguel et al., 1997)
Amman (Jordan)	271	351	139	29	1.9	-	-	32	66	(Al-Momani, 2009)
Abadan (Iran)	59	288	112.97	50	0.52	7	36	8	57	This Study

Table 4

			Er		
Element	**Min**	**Max**	**Mean**	**SD**	**Skewness**
Pb	5	87	20	14	4
Zn	4	23	9	5	1
Cu	5	169	19	29	5
Cr	2	4	3	0.48	0.35
Cd	65	113	78	11	1
Ni	9	22	14	2	0.69
V	0.93	2	1	0.13	0.71
As	8	61	15	11	3

Table 5

	RI			Number of sample	Number of sample			
	Min	**Max**	**Mean**	**Number of sample**	**Low risk**	**Moderate risk**	**Considerable risk**	**High risk**
Street dust	115	330	159	30	17(% 56.66)	12(% 40)	1(% 3.34)	0

Table 6

Element	Pb	Zn	Cu	Cr	Cd	Ni	V	As	Co
Pb	1								
Zn	0.680**	1							
Cu	0.861**	0.840**	1						
Cr	0.366*	0.546**	0.511**	1					
Cd	0.556**	0.525**	0.508**	0.021	1				
Ni	-0.139	0.052	-0.020	0.699**	-0.300	1			
V	0.054	0.271	0.168	0.785**	-0.149	0.826**	1		
As	0.466**	0.513**	0.465**	0.385*	0.508**	-0.004	0.308	1	
Co	0.030	0.262	0.121	0.789**	-0.088	0.849**	0.900**	0.261	1

** Correlation is significant at the 0.01 level (2-tailed).
* Correlation is significant at the 0.05 level (2-tailed).

Table 7

Element	Component			Communities
	1	2	3	
Pb	-0.037	**0.717**	0.329	0.623
Zn	0.249	**0.869**	0.248	0.878
Cu	0.157	**0.864**	-0.134	0.789
Cr	**0.804**	0.345	0.112	0.778
Cd	-0.155	0.246	**0.892**	0.881
Ni	**0.932**	0.050	-0.156	0.895
V	**0.939**	0.112	-0.046	0.897
As	0.088	0.040	**0.884**	0.790
Co	**0.973**	0.009	0.044	0.949
Initial Eigenvalue	3.815	2.428	1.237	
Total variance %	42.389	26.980	13.744	
Cumulative %	42.389	69.368	83.113	

Table 8

Compounds	Aromatic ring	TEF	Mean	Min	Max	S.D
Acenaphthylene (Acy)	3	0.001	84	19	396	68
Fluorene (Fl)	3	0.001	167	29	399	81
Phenanthrene (Phe)	3	0.001	382	69	875	184
Anthracene (Ant)	3	0.001	104	11	796	137
Pyrene (Pyr)	4	0.001	753	169	3541	644
Benzo(a)anthracene (BaA)	4	0.100	102	8	1118	198
Chrysene (Chr)	4	0.010	278	42	1267	227
Benzo(b)fluoranthene (B(b)F)	5	0.100	153	19	1166	210
Benzo(k)fluoranthene (B(k)F)	5	0.100	85	7	602	112
Benzo(a)pyrene (BaP)	5	1.000	61	4	496	101
Indeno(1,2,3-cd)pyrene (IcdP)	5	0.100	155	6	576	197
Dibenzo(a,h)anthracene (DBA)	6	1.000	46	13	79	33
Benzo(ghi)pyrylene (BghiP)	6	0.010	66	3	456	102
3 ring %			30	32	21	21
4 ring %			47	55	50	47
5 ring %			18.6	9	24	27
6 ring %			4.61	4	5	6
∑PAHs			2436	400	11766	2295
LMW PAHs			737	128	2467	471
HMW PAHs			1699	272	9300	1825
COMPAHs/Σ13PAHs			0.74	0.77	0.80	0.73
NCANPAHs/Σ13PAHs			0.72	0.60	0.80	0.63
CANPAHs/Σ13PAHs			0.28	0.20	0.40	0.37
TEQ			162	22.	951	211
TEQ/ΣPAHs			0.07	0.06	0.08	0.09

ΣPAHs: Sum of individual mass content of detected PAHs .

LMWPAHs: Sum of low molecular weight 3 ring PAHs.
HMWPAHs: Sum of high molecular weight 4–6 ring PAHs.
COMPAHs: Sum of major combustion derived PAH content.
CANPAHS: Sum of carcinogenetic PAHs.
NCANPAHs: Sum of non-carcinogenetic PAHs.
TEF: Toxic equivalence factors (data from (Collins et al., 1998; Tsai et al., 2004).
TEQ: Toxic equivalency concentration.

Table 9

	Acy	Fl	Phe	Ant	Pyr	BaA	Chr	B(b)F	B(k)f	BaP	IcdP	DBA	BghiP	{PAH	LMW PAHs	HMW PAHs	TEQ
Acy	1.000																
Fl	0.733**	1.000															
Phe	0.407*	0.749**	1.000														
Ant	0.312	0.526**	0.649**	1.000													
Pyr	0.212	0.387*	0.696**	0.603**	1.000												
BaA	0.182	0.366*	0.614**	0.685**	0.838**	1.000											
Chr	0.170	0.242	0.638**	0.539**	0.830**	0.777**	1.000										
B(b)F	0.157	0.246	0.611**	0.561**	0.838**	0.871**	0.940**	1.000									
B(k)f	0.075	0.223	0.595**	0.632**	0.846**	0.840**	0.853**	0.913**	1.000								
BaP	0.162	0.340	0.552**	0.506**	0.783**	0.900**	0.769**	0.878**	0.836**	1.000							
IcdP	-0.021	0.101	0.414*	0.348	0.575**	0.598**	0.631**	0.655**	0.664**	0.652**	1.000						
DBA	0.116	0.256	0.433*	0.387*	0.433*	0.433*	0.433*	0.433*	0.433*	0.434*	0.619**	1.000					
BghiP	-0.056	0.211	0.451*	0.377*	0.635**	0.653**	0.673**	0.773**	0.679**	0.798**	0.527**	0.441*	1.000				
ΣPAHs	.0319	0.531**	0.777**	0.782**	0.915**	0.800**	0.822**	0.818**	0.863**	0.755**	0.596**	0.433*	0.594**	1.000			
LMW PAHs	0.548**	0.770**	0.826**	0.815**	0.580**	0.464**	0.484**	0.429*	0.515**	0.398*	0.331	0.432*	0.297	0.786**	1.000		
HMW PAHs	0.141	0.309	0.646**	0.603**	0.966**	0.895**	0.900**	0.928**	0.913**	0.875**	0.629**	0.433*	0.728**	0.906**	0.510**	1.000	
TEQ	0.163	0.327	0.584**	0.587**	0.836**	0.927**	0.834**	0.927**	0.899**	0.980**	0.658**	0.433*	0.782**	0.823**	0.453*	0.923**	1.000

** Correlation is significant at the 0.01 level (2-tailed).

* Correlation is significant at the 0.05 level (2-tailed).

Table 10

	Factor 1	Factor 2	Factor 3
Acy	-0.123	**0.783**	-0.044
Fl	0.137	**0.947**	-0.059
Phe	0.355	**0.635**	0.038
Ant	0.116	0.004	**0.991**
Pyr	**0.922**	0.229	0.110
BaA	**0.982**	0.055	0.079
Chr	**0.935**	0.197	0.104
B(b)F	**0.983**	0.107	0.072
B(k)f	**0.970**	0.137	0.124
BaP	**0.984**	0.110	0.049
IcdP	**0.978**	0.051	0.071
DBA	**0.953**	-0.011	0.077
BghiP	**0.955**	0.085	0.044
Total variance %	63.606	16.365	7.501
Cumulative %	63.606	79.971	87.472

Table 11

Compounds	TEF	NCs	MPCs	RQ (NCs)	RQ (MPCs)
		($\mu g\ kg^{-1}$)			
Acy	0.001	1.2	120	70	0.7
Fl	0.001	1.2	120	139	1.4
Phe	0.001	5.1	510	75	0.7
Ant	0.001	1.2	120	87	0.9
Pyr	0.001	1.2	120	628	6.3
BaA	0.100	2.5	250	41	0.4
Chr	0.010	107	10700	3	0.0
B(b)F	0.100	2.5	250	61	0.6
B(k)f	0.100	24	2400	4	0.0
BaP	1.000	2.6	260	23	0.2
IcdP	0.100	59	5900	3	0.0
DBA	1.000	2.6	260	18	0.2
BghiP	0.010	75	7500	0.9	0.0
ΣPAHs				1151	7.7

Table 12

Exposure pathways	Child				Adult			
	$ILCRs_{ing}$	$ILCRs_{der}$	$ILCRs_{inh}$	Cancer risk	$ILCRs_{ing}$	$ILCRs_{der}$	$ILCRs_{inh}$	Cancer risk
Mean	3.98E-04	4.97E-04	7.70E-09	8.95E-04	3.11E-04	5.52E-04	2.41E-08	8.64E-04
Min	5.43E-05	6.77E-05	1.10E-09	1.22E-04	4.24E-05	7.53E-05	3.30E-09	1.17E-04
Max	2.34E-03	2.91E-03	4.54E-08	5.26E-03	1.82E-03	3.24E-03	1.41E-07	5.07E-03