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Contamination Level, Source Identification and Risk Assessment of Potentially Toxic Elements (PTEs) and Polycyclic Aromatic Hydrocarbons (PAHs) in Street Dust of an Important Commercial Center in Iran

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Abstract

The geochemical nature and health hazards of potentially toxic elements (PTEs) and polycyclic aromatic hydrocarbons (PAHs) in the street dust of Bandar Abbas, Iran, are investigated in this study based on 27 street dust samples. Mean concentrations of Cu, Pb, Zn, As, Sb, and Hg revealed elevated concentrations as compared to the world soil average. Calculated enrichment factors (EFs) indicated that there is very high contamination in dust particles owing to anthropogenic emissions. Two main sources of PTEs are traffic emissions (Cu, Pb, Zn, Co, Mn, Fe, As, Cd, Sb, and Hg) and resuspended soil particles (Al, Ti, Ni, and Cr). Statistical analysis shows that Al, Mn, Ni, Ti, Cr, Fe, and Co are geogenic, whereas PAHs are mainly derived from traffic emissions. Values of incremental lifetime cancer risk (ILCR), as derived from a modified model of the United State Environmental Protection Agency (USEPA), indicate that Bandar Abbas residents are potentially exposed to high cancer risk, especially via dust ingestion and dermal contact, whereas the level of hazard index (HI), hazard quotients (HQ), and cancer risk associated with exposure to the elements in street dust fall lower than threshold values representative of health risks.

Keywords

Street dust; Potentially toxic elements (PTEs); Polycyclic aromatic hydrocarbons (PAHs); Bandar Abbas; health risk assessment

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Compliance with Ethical Standards

Conflict of Interest The authors declare that they have no conflict of interest.

Introduction

Streets represent an essential component of the urban landscape and are significant sources of contaminants (Zhao et al. 2009). Urban surfaces receive deposits of particulate matter issued from sources, such as vehicular traffic, industrial activity, domestic heating, and waste incineration (Selinus et al., 2005). Street dust is the result of those emissions depositing on paved streets. In addition, particles that settle on the street from both nearby and distant sources can be subsequently resuspended by mechanical lifting, such as from vehicles or wind, as has been demonstrated in a number of areas, such as the Sonoran Desert (Prabhakar et al., 2014) and Mexico City (Mora et al., 2017). Consequently, street dust particles in urban areas are both a sink and source of contaminants such as potentially toxic elements (PTEs) and polycyclic aromatic hydrocarbons (PAHs) (Charlesworth et al. 2003; Zhao et al. 2006; Zhang and Wang 2009; Soltani et al. 2015).

Street dust is comprised of a complex mixture of constituents including dust and emissions stemming from tire and brake pad wear, vehicular exhaust, spillages and leaks from motor vehicles, erosion material from the street surface, and even remnants of animals and plants. Consequently, street dust particles are often contaminated with a wide range of toxic elements and organic compounds, including PAHs (Takada et al. 1991; Rogge et al. 1993; Pereira Netto et al. 2006; Zhang et al. 2008a; Dong and Lee 2009; Mostafa et al. 2009). The concentrations and elemental composition in street dust are sensitive to the characteristics of specific regions. PTEs have long half-lives in the human body because they are non-biodegradable and are slowly eliminated; therefore, PTEs in urban areas have been a subject of great concern (Keshavarzi et al. 2015b; Al-Khashman and Shawabkeh 2009). Ingestion, inhalation, and dermal adsorption are the most important ways that street dust can be absorbed by humans (Wei et al. 2010). Health-related properties such as carcinogenicity, genotoxicity, immunotoxicity, and reproductive toxicity reach especially concerning levels upon exposure to PAHs with two or more aromatic rings (Chrysikou et al., 2009).

In terms of natural sources, PAHs are produced by volcanic activity, microbial synthesis, and synthesis in terrestrial vegetation. These sources are small in comparison with anthropogenic sources, such as biomass burning, petroleum products, vehicular exhaust, lubricating oils, petroleum refining, materials weathered from street surfaces, asphalt pavement, tire particles, and construction materials. It is well documented that in urban areas, anthropogenic sources are responsible for elevated levels of PAHs in street dust (Zhang et al. 2015; Liu et al. 2007; Aryal et al. 2006).

In developing countries like Iran, there are enormous problems associated with the pollution leading to contamination of street dust due to anthropogenic activities. Bandar Abbas in the Hormozgan province has a hot desert climate and is located in the south of Iran. The city is situated on flat ground with an average altitude of 9 m above sea level. The annual rainfall accumulation is around 170 mm with an average relative humidity of 65%. The annual average number of days with dust storms is 86, with the wind direction predominantly being southerly (www.chaharmahalmet.ir).

This paper focuses on the following: (1) analysis of street dust to characterize the concentration and possible source(s) of selected PTEs and PAHs; and (2) evaluate the human health risk of PTEs and PAHs to both children and adults living in Bandar Abbas.

Material and Methods

Sampling, Sample Preparation and Analytical Procedures

Twenty-seven samples (each weighting ~500 g) collected from each sampling site were homogenously mixed to form a composite sample (Fig. 1). Dust samples were collected from street surfaces and adjacent to street curbs in different locations of Bandar Abbas during January 2016. Sampling involved gently sweeping an area of about 2 m^2 using a plastic hand broom with the contents transferred to a clean, self-sealed polyethylene bag. In the laboratory, after air-drying, the coarse impurities of the samples such as stones, cigarette butts, plastics, and leaves were removed and the rest was ground with an agate mortar and pestle to be passed through a 63 mm (220 mesh) nylon sieve. Finally, PTEs were analyzed using inductively coupled plasma mass spectrometry (ICP-MS) at the accredited Acme Analytical Laboratory in Canada, using the digestion method of Qi and Gregoire (2000).

In addition to PTEs analysis, seventeen street dust samples were also collected and preserved in a solvent-cleaned glass jar for PAH analysis. The samples were stored in an ice-chest at 4 °C and transferred to the laboratory of Isfahan University of Technology where the samples were prepared for PAH analysis using gas chromatography/mass spectrometry (GC/MS). The samples were analyzed according to EPA 3550B (extraction), EPA 3630 C (clean up), and EPA 8310 (determination). The average recoveries based on surrogate deuterated PAHs in dust were ~88–95% for the 16 measured PAHs. The detection limit (DL) of PAHs for dust ranged from 0.05–0.4 μg/kg and precision was 4–8%.

Pollution Assessment

A number of geochemical methods have been used to quantify the degree of PTE enrichment in dust samples. In the present study, enrichment factors (EFs) were calculated to assess PTE contamination levels. EF can also help discriminate the anthropogenic sources from natural ones. For each element (i), EF is calculated as follows:

$$\mathrm{EF} = \frac{\left(\frac{C_{\mathrm{i}}}{C_{\mathrm{ref}}}\right)_{\mathrm{sample}}}{\left(\frac{C_{\mathrm{i}}}{C_{\mathrm{ref}}}\right)_{\mathrm{background}}} \quad (1)$$

Where (C_i/C_{ref}) is the ratio of concentration of element i to the concentration of a reference element in the sample vs. background. A reference element is characterized by low occurrence of variability with minimal anthropogenic influence, with some commonly used elements being: Al, Fe, Ti, Sc, Si, and Zr (Najmeddin et al. 2017; Han et al. 2006, Kartal et al. 2006; Meza-Figueroa et al. 2007; Turner and Simmonds 2006; Sorooshian et al., 2012). The EF was split into five classes as follows based on its values (Birch and Olmos, 2008;

Yongming et al., 2006): deficiency to minimal enrichment (<2), moderate enrichment (2–5), significant enrichment (5–20), very high enrichment (20–40), and extremely high enrichment (>40).

Source Apportionment Analysis

To identify the association among PTEs and PAH species in street dust and their possible sources, Spearman's correlation coefficients were calculated and principal component analysis (PCA) and cluster analysis (CA) were performed using the commercial statistics software package SPSS version 19.0 for Windows (Keshavarzi et al. 2015a). PCA and CA were applied to distinguish between hypothetical source of PTEs and PAHs (natural or anthropogenic sources). The aim of PCA is to reduce a larger set of variables into a smaller set of variables, which account for most of the variance in the original variables. Also, CA is the process of grouping a set of objects in such a way that objects in the same group (called a cluster) are more similar (source or physicochemical behavior) to each other than to those in other groups (clusters).

Exposure and Risk Assessment

A modified model of the United States Environmental Protection Agency (USEPA) is used in this study to assess exposure risk of adults or children to PTEs in street dust (US EPA 1996). Also, the Dutch National Institute of Public Health and Environmental Protection (Van den Berg 1995) was relied upon for the definition of guidelines or screening levels of contaminants in street dust in different urban exposure scenarios. The doses received through the three pathways (i.e., ingestion (ing), inhalation (inh), and dermal contact) of individual elements and PAHs were calculated following the guidance of past work (USEPA 1989; Cao et al., 2017). The model approach used here has several weaknesses that include: (i) not considering the particle size of dust; (ii) values for ingestion rate (Ing_R), skin adherence factor (SL), and inhalation rate (Inh_R) were based on exposure for a whole day (24 h), however it does provide a conservatively more protective assessment, but in fact it is unrealistically long; (iii) the recommended particle emission factor (PEF) value is 1.36×10^9 m^3 kg^{-1}, which can be understood as the space occupied by 1 kg of dust. In the current study, we analyzed street dust with particle sizes less than 63 μm and although we can calculate the exposure dose via inhalation (ADDinh) the value of ADDinh is negligible, therefore, we did not include it during calculation.

$$\mathrm{ADD_{ing}} = \frac{C \times \left[\mathrm{IngR} \times f\right] \times \left[\mathrm{EF} \times \mathrm{ED} \times 10^{-6}\right] \times \mathrm{AR}}{\mathrm{BW} \times \mathrm{AT}} \quad (2)$$

$$\mathrm{ADD_{dem}} = \frac{C \times \left[\mathrm{SA} \times \mathrm{SL} \times f\right] \times \left[\mathrm{ED} \times \mathrm{EF} \times 10^{-6}\right] \times \mathrm{ABS}}{\mathrm{BW} \times \mathrm{AT}} \quad (3)$$

The lifetime average daily dose for carcinogens is calculated as follows:

$$\mathrm{LADD}\left(\mathrm{mg\ kg^{-1}day^{-1}}\right)=\frac{C\left(\mathrm{mg\ kg^{-1}}\right)\times \mathrm{EF}}{\mathrm{AT}}\times\left(\frac{\mathrm{CR_{child}}\times \mathrm{ED_{child}}}{\mathrm{BW_{child}}}\times\frac{\mathrm{CR_{adult}}\times \mathrm{ED_{adult}}}{\mathrm{BW_{adult}}}\right) \quad (4)$$

For related exposure factors and their variables for the modified models, the reader is referred to the work of Cao et al. (2017) and Abbasi et al. (2017).

Result and Discussion

Trace Element Concentrations

Descriptive statistics of PTE concentrations in the street dust of Bandar Abbas, as well as background values of world-average soils (Kabata-Pendias and Pendias 2001) and upper crust content (UCC) (Rudnick and Gao 2003), are presented in Table 1. The mean concentrations of Sb, As, Hg, Cu, Pb, Zn, and Ni in street dust samples were 3.79, 13.95, 0.2, 203.76, 165.59, 214.13, and 71 mg kg^{-1}, respectively, which are found to be substantially higher than world-average soil values and UCC. The normality of the data was checked using the Shapiro–Wilk test. The results show that most of the data are non-normally distributed (p-value < 0.05).

Table 2 indicates that all elements (except for Ni, Cr, Ti, and Al) are positively skewed. Furthermore, variation coefficients (VCs = standard deviation/mean) are high for all elements except for Ni, Cr, Ti, Mn, Co and Al. Elements dominated by a natural source are expected to have low VCs, while those affected by anthropogenic sources and with high non-homogeneity, typically exhibit high VCs (Yongming et al., 2006).

PTE concentrations in Bandar Abbas street dust samples are compared with data reported for other cities in the world in Table 2. It should be noted that comparison of mean concentrations of PTEs in street dust in different urban environments is a common practice, despite the fact that there are no universally accepted sampling and analytical procedures for geochemical studies of urban deposited materials. Also, concentrations of PTEs in street dust particles vary considerably among cities depending on the local climate condition, wind patterns, technologies, as well as density of traffic and industrial activities. Generally, Pb is the element of most concern in environmental heavy metal pollution. The mean and median level of Pb in the present study is 165.59 and 66.36 mg kg^{-1}, respectively. Lead levels in this work are consistent with findings in Oslo (Norway) and are slightly higher than those found in Bushehr (Iran), Accra (Ghana), Seoul (Korea), and are lower than those found in Tehran (Iran), London (U.K.), Madrid (Spain), and Kavala (Greece). Concentrations of all elements studied (except Zn) in Bandar Abbas city are higher than Bushehr city. It is worth noting that the studies in Bushehr and Bandar Abbas cities were performed under similar conditions associated with climate, weather, and sampling methods. The main reason for the concentration difference of PTEs in both cities is anthropogenic activity, especially stemming from heavy traffic. More specifically, the population in Bandar Abbas city

(680,366 people), and thus traffic load, is higher than Bushehr city (298,945 people). Thus, traffic load and population are important factors that affect contamination of dust since the chief source of Pb in the study region is emissions from vehicular exhaust owing to usage of leaded gasoline in Bandar Abbas in past years. Also, the mean Cu concentration is much higher than for all other cities, except for Tehran. The mean concentration of Zn in Bandar Abbas street dust is only higher than dust sampled in Ottawa, Canada. Existence of Cu in street dust could be derived from engine wear of automobiles while attrition of automobile tires and lubricating oils are possible sources of Zn and Cd (Akhter and Madany 1993).

PTEs Contamination in Street Dust

EFs of PTEs are calculated for each element relative to the background value reported for UCC. Bandar Abbas street dust has not been characterized prior to the current study, thus there are no reported historical background elemental concentrations for the area. In this study, Al is used as a reference element. The box-plots and GIS maps of EF for street dust from Bandar Abbas are provided in Figs. 2 and 3, respectively. Although median values in skewed datasets are more reliable to represent the dataset, mean values of parameters (e.g., concentration, EF) were used to compare the data with other similar studies. The mean EF values displayed the following decreasing trend: Sb > As > Pb > Cu > Cd > Hg > Ni > Zn > Cr > Co > Mn > Fe > Ti. EF values for Ti, Fe, Mn, and Co ranged from 1 to 10, further confirming that they mainly originate from natural sources. On the other hand, other elements, with maximum EFs much higher than 10, were considered believed to have originated mainly from anthropogenic sources (Liu et al. 2003).

Source Identification of PTEs

Statistical analyses (i.e., Kolmogorov–Smirnova and Shapiro–Wilk tests, skewness, and standard deviation) reveal that the PTE concentration data are non-normally distributed (Table 1). The Spearman correlation coefficient was calculated using SPSS 19.0 to examine the relationship between the analyzed PTEs. Significant correlations ($r > 0.5$) were found between ΣPAHs and Pb, Zn, Fe, As, Cd, and Sb. High correlations were also observed between Cu and Pb, Zn, Fe, As, Cd, and Sb, in addition to between Al and Ti, Ni, and Cr. ΣPAH values showed very weak positive correlations with Al, Ti, Ni, and Cr (−0.32 to 0.07). A strong correlation was not found between Al, Ti, Cr, and Ni with other elements. Anthropogenic activities are the main sources of ΣPAHs in the street dust samples. Consequently, high correlations between elements and ΣPAHs are mostly due to their anthropogenic origin (Table 3).

PCA and CA were used to more accurately identify the source of the pollutants. Before using PCA, Bartlett and KMO tests show that the data are suitable for PCA test. Figures 4 and 5 display the component plot in rotated space with varimax rotation, as well as the eigenvalues. The results indicate that there were two components with eigenvalues higher than one explaining 85.2% of the total variance. The first factor includes Cu, Zn, As, Cd, Pb, Fe, Mn, Hg, Co, and Sb, while the second factor includes Al, Ti, Cr, and Ni. The results of CA are in accordance with PCA (Figs. 4 and 5). CA was applied to standardized bulk concentration data using between-groups linkage method, with z-score distances as the criterion for forming clusters of elements. CA, PCA, VCs, and EF results suggest that Cu,

Zn, Sb, As, Cd, Pb, and Hg likely originated from vehicular traffic (anthropogenic origin), whereas Al, Mn, Ni, Ti, Cr, Fe, and Co have a natural origin (e.g., local soils). Hg exhibits different physicochemical behavior as compared to other anthropogenic elements, hence the element is located away from the other anthropogenic elements in CA (Fig. 5). Fe, Mn, and Co are elements with mixed sources (e.g., anthropogenic and natural origin). The variation coefficients of Mn and Co were lower than 0.4. Although these elements have low EF values, they are associated with anthropogenic elements in the PCA and CA. Outliers in Fig. 2 include the sampling sites H24, H25, and H26. These samples are located near a transit road in which trucks transport Fe-rich minerals. Also, Figs. 2 and 3 show that concentrations of anthropogenic elements such as Pb, Cu, Zn, Sb, and As increased in H24, H25, and H26. Thus, this confirms that the main anthropogenic sources are automobiles and especially trucks (e.g., brakes or exhaust or part wear, etc.), which is in accordance with Abbasi et al. (2017). In addition, the major source of Pb associated with street dust is fuel additives from automobiles (De Miguel et al., 1997). The use of Pb in gasoline was banned many years ago; however, since Pb is an enduring element and has little mobility, it can remain in the environment for a long time. The use of Pb in exterior paints, industries, and graphics wastes may also be an important source of Pb in dust, especially in low traffic areas (Faiz et al., 2009). On the other hand, combustion of fossil fuels and abrasion of vehicle tires can continue to release Cd (Kabata-Pendias and Mukherjee 2007). According to Kabata-Pendias and Mukherjee (2007), the potential sources of Cu and Zn include abrasion of automobile tires, car brake pads, car wash, and street paving. It is reasonable to consider Cu, Zn, As, Cd, Pb, Fe, Mn, Hg, Co, and Sb as anthropogenic elements released from transportation sources (Kabata-Pendias and Mukherjee 2007) in the current study. Generally, according to region conditions (such as wind speed, rainfall, temperature, human activity and etc.), street dust is both a source and a reservoir of contaminants. Thus, when street dust is considered as a source of contamination, the majority of street dust includes geogenic elements (e.g., Al, Ti and etc.). In fact, naturally occurring materials are a critical component of street dust and often represent the majority of measured mass of the street dust.

Health Risk Assessment of PTEs Exposure From Urban Street Dust

The results of the risk assessment (HQ and HI) are shown in Table 4. Slope factors (SLF) and oral reference doses (RfDo) are listed in Table 4. The hazard index (HI) is quantified by summing up the hazard quotients (HQ) of each PTE for each pathway. HI values larger than unity reflect a significantly adverse non-carcinogenic effect (USEPA 2001).

There are three main routes by which elements derived from anthropogenic activities can enter the human body: ingestion, inhalation, and direct dermal contact (Davies and Mundalamo 2010). As for the three exposure pathways, ingestion was the dominant way for both adult and children. Generally, children are exposed to more street dust elements than adults via each of the three pathways. They also commonly have more frequent hand-to-mouth activities. Meanwhile, for adults, the daily outdoor activities such as eating food outside, wiping sweat, or dirty hands or faces before eating or drinking may also lead to the possibility of ingestion of trace elements in dust impacts on the body (Huiming Li et al. 2013). HI values for the studied elements were all lower than the safe level of 1, indicating little adverse health risks from street dust exposure. Similar results were obtained in

previous studies examining urban dust (Abbasi et al., 2017; Zhang et al. 2013; Huang et al. 2014; Li et al. 2013; Ferreira-Baptista and De Miguel 2005). The level of cancer risk associated with exposure to the elements in studied samples falls lower than the range of threshold values (10^{-4}–10^{-6}) which environmental and regulatory agencies consider as acceptable.

PAH Concentrations

The 16 targeted PAHs were detected in street dust samples from Bandar Abbas city. The summary statistics for PAHs concentrations are shown in Table 5. In general, total PAH (ΣPAHs) concentrations ranged between 73.08 and 1339 μg kg^{-1}, with an average of 362.33 μg kg^{-1}. The mean concentration of lower molecular weight PAHs (LMW PAHs) with 2 or 3 aromatic rings was 95.53 μg kg^{-1}, accounting for 25.81% of the total PAH concentration. The mean concentration of higher molecular weight PAHs (HMW PAHs) with four to six rings was 266.8 μg kg^{-1}, accounting for 74.19% of the total PAH concentration. The inverse distance weighing (IDW) interpolation analysis, based on ΣPAHs, was applied to visualize the spatial distribution of data using ArcGIS 10.3 software in Fig. 6. The highest ΣPAH concentration was found in the pavement surface dust from the gas station (B11) where the high concentrations of PAHs may have resulted from deposition of emissions from the slow moving vehicles and leakage of petroleum products (petrol and diesel) (Dong and Lee 2009). Leakage of petroleum products promote increased LMW PAH concentrations (Mai et al. 2003), which was evident at the B11 sampling site. Also, the lowest ΣPAH concentration was found in residential areas (such as sampling site of B13) with low traffic load and relatively lower amounts of anthropogenic activity. This study revealed that the highest average concentration for a PAH species in Bandar Abbas street dust belongs to Pyr with 49.18 μg kg^{-1} followed by Phe, Flu, Chr, B[e]p, B[k]f, Db[ah]a, B[a]p, B [ghi]p, B[a]a, Ant, Np, Ind, B[b]f, Fl, and Ace. Combustion PAHs (ComPAHs), the sum of Flu, Pyr, BaA, Chr, B(b + k)F and BaP, (Rogge et al. 1993; Rajput and Lakhani 2009), comprise a significant proportion of the total concentration of PAHs in the measured street dust, ranging from 0.57 to 0.78 μg kg^{-1}. The mean ratio of carcinogenic PAHs (according to Rajput and Lakhani, 2009, include Nap, B[a] a, Chry, B[e]p, B[b]F, B[k]f, B[a]p, Db[ah]a and Indene) to total PAHs (CANPAHs/ΣPAHs) is 0.48 μg kg^{-1} and ranges between 0.3 to 0.68 μg kg^{-1}. Also, the ratio of non-carcinogenic PAHs to total PAHs (NCANPAHs/ΣPAHs) approximately equals CANPAHs/ΣPAHs, indicating that carcinogenetic PAHs are abundant in this area.

One way of classifying PAHs is the number of aromatic rings. The compositions of PAHs in the street dust samples are shown in Fig. 7. The average concentrations of LMW PAHs and HMW PAHs accounted for 12.36–40.06% and 59.94–87.64% of ΣPAHs, respectively. The results indicate that the dominant components of PAHs in Bandar Abbas street dust were HMW PAHs, particularly four-ring PAHs. This might be due to the tendency of higher molecular weight compounds to adhere to street dust (Wang et al. 2011). Also, PAHs with more than four rings generally come from pyrogenic (petroleum fuels combustion) sources in cities (Dahle et al. 2003) and generally come from motor vehicles (Halek et al., 2008).

The mean concentration of PAHs in Bushehr city (Keshavarzi et al., 2017) are higher than Bandar Abbas city (with the exception of PTEs). PAH levels were especially high at two

stations (B13 and B11) in Bushehr city (Keshavarzi et al., 2017). Also, the traffic load and population of Ulsan (1,049,177), a coastal city in Korea, are more than Bandar Abbas city. Consequently, the concentration of total PAHs in Ulsan street dust (45,750–112,130 μg kg $^{-1}$) (Dong et al. 2007) are higher than those measured in Bandar Abbas.

Source Identification of PAHs

Shapiro–Wilk, skewness, and standard deviation results showed that the data is non-normally distributed. Results of PCA and correlation coefficients among PAH compounds based on Spearman's correlation analysis agree with each other (Table 3). PCA analysis was used to represent the total variability of the original PAH data with a minimum number of factors, and estimate the chemical source responsible for each factor (Larsen and Baker 2003). In the present study, PAH data were normalized by Box-Cox Transformation and Johnson normalization methods. PCA was conducted with varimax rotation.

The results are presented in Table 6, where three PCs are identified that represent different source categories or physicochemical behavior. The first PC was dominated by 5–6 ring PAHs, including B[e]p, B[b]f, B[k]f, B[a]p, Db[ah]a, B [ghi]p, and Ind, which are identified as markers of gasoline emissions (Duval and Friedlander 1981; Khalili et al. 1995; Park et al. 2002; Guo et al. 2003) and finally representing vehicle-related sources and oil combustion (Larsen and Baker 2003; Dong and Lee 2009; Harrison et al. 1996). The second PC mostly associated with the LMW PAHs and displays high loading values of Np, Ace, Fl, Phe, Ant, Pyr, B[a]a, and Chr. These PAH species have low melting and boiling points among the 16 PAH congeners. For example, Np and Ace have the lowest and second lowest melting (80 °C and 108 °C, respectively) and boiling points (209 °C and 252 °C, respectively) (Wang 2007). The high loading values of these PAHs indicates that this factor represents wood combustion-related sources (Fang et al. 2004). Flu appears in PC3 due to its high stability among of 16 PAHs (Chung et al. 2007).

PAH ratios are commonly used to determine PAH sources, clarify the samples by location, and estimate the importance of combustion and petroleum-derived PAHs (Yunker et al. 2002). For example, HMW-PAHs were abundantly generated at high temperature and combustion-derived, while the petroleum-derived contain relatively high abundances of LMW-PAHs (Mai et al., 2003). Thus, the ratio of LMW/HMW could be used as an index to apportion the anthropogenic sources of PAHs (Fernandes et al. 1997) where it can be separated into two categories (i.e.,>1 and <1), indicating petrogenic and pyrogenic sources, respectively (Zhang et al. 2008b). The ratios of LMW/ HMW in street dust samples from Bandar Abbas city are lower than 1 (Fig. 7), indicating the predominant pyrogenic sources of PAHs.

Values of the Phe/Ant ratio <10 typically indicate that PAHs mainly come from pyrogenic origin; in contrast, ratios >15 indicate petrogenic sources (Takada et al. 1991). This ratio was used to identify the sources of PAHs in the present study. The results showed that Phe/Ant was <10, suggesting that the main sources were pyrogenic sources in the study region (Fig. 8).

The values of BaA/(BaA + Chr) and Ind/(Ind + B(ghi)P) are summarized in Fig. 8. These ratios reveal that pyrogenic sources are predominant in our street dust samples, characterizing the vehicle exhaust emissions (especially gasoline emission) (Yunker et al. 2002; Gogou et al. 1996; Khalili et al. 1995).

Carcinogenic Risk Assessments of PAHs

Total cancer risk and incremental lifetime cancer risk (ILCR) for human health risk from PAHs exposure was calculated (Table 7). ILCR is used as a representative to identify the age-specific potential cancer risks (children and adults) of human exposure to environmental PAH pollution sources (Peng et al. 2011; Chen and Liao 2006). Generally, the acceptable risk range for carcinogens is set to 10^{-6} –10^{-4} by the USEPA (USEPA 2001) and for risks below 10^{-6} do not require further action, while risks above 10^{-4} are considered to be of concern and require additional action to reduce the exposure and resulting risk (USEPA 2008).

In the current study, average, maximum, and minimum concentrations of the 16 investigated PAHs were used in the calculations of risk assessments. The mean cancer risk via the inhalation pathway is negligible for adults and children. Their levels via dermal contact and ingestion pathway are 2.14×10^{-5} and 1.43×10^{-7} in children, and 2.99×10^{-6} and 1.53×10^{-8} in adults, respectively (Table 7). The risk value of ingestion pathway for adults is slightly lower than the children. The average value of total cancer risk for children (2.16×10^{-5}) and adults (3.00×10^{-6}) were calculated by the sum of the ILCRs through two exposure pathways, which was higher than 10^{-6} for adults and lower than 10^{-6} for children. Generally, children are the most susceptible subpopulation, because they have lower body weight relative to adults and PAH intake (mg/kg-body-weight/day) is greater. Furthermore, early growth of organs, immune and nervous systems can increase sensitivity of children (Maertens et al. 2008). Also, the contaminated dust can be readily ingested by children via their hand-to-mouth activity (Meza-Figueroa et al. 2007), and thus the carcinogenic and non-carcinogenic hazard health risk for children exposed to street dust PAHs is greater than that of adults. Finally, it should be noted that human health risk assessment models have several uncertainties and also there are likely unknown factors that influence human health. Also, in addition to PTEs and PAHs, the presence of other contaminants and their synergistic or antagonistic effects are not considered. For example, microplastics along with other contaminations can enter the human body through the food chain (Abbasi et al. 2017). Thus, the result of the models (for both of the PTEs and PAHs), are not fully comprehensive and have limitations in terms of being validated via field measurements.

Management and Planning

Street dust is composed of particles that arise from local soils, traffic emission (vehicle exhaust particles, tire wear particles, weathered street surface particles, brake lining wear particles), street pavements, microrubbers, and microplastics (Abbasi et al., 2017; Yeung et al. 2003a, b). Due to the high levels of significant organic and inorganic contaminants in street dust, there is a dual potential negative impact to human and ecological health from wet deposition (and surface storm water runoff) and air quality (Calvillo et al., 2015). Street cleaning is listed as a best management practice for enhancing storm water quality by the

EPA, data are limited on the critical parameters (technology, environment, usage), which determine the effectiveness of any street cleaning program. Therefore, street cleaning program and planning related to street dust in urban areas is important. It is also clear from the available data that local conditions, climate, and specific needs are critical determinants of the ideal street sweeping strategy (technology, frequency, speed, targeted areas, etc.) (Calvillo et al., 2015).

Street cleaning in relation to environmental risk or human health has been evaluated in only a few studies (Zhao et al., 2009, 2008). The California Storm Water Quality Association lists several suggested protocols for street sweeping and cleaning (CSQA 2003). Few studies have compared the effectiveness of street cleaning technologies to remove street dust (Amato et al., 2010; Kang et al., 2009). A number of different street cleaning technologies and practices for improvement of air quality are available to manage and collect street dust, including flushing, mechanical broom, vacuum, and regenerative air systems (Whitacre, 2016). Understanding the levels, distribution, and sources of PTEs and PAHs in street dust can aid environmental managers and facilitate the supervision of air quality. The results of the present work indicate that street dust within Bandar Abbas contain high concentrations of PTEs and PAHs reflecting the contribution of vehicular traffic and the industrial activities. However, such studies are either scanty or undocumented in Iran, a country experiencing rapid, but unplanned urbanization, increased vehicular traffic, fuel combustion, and construction. Unfortunately, Bandar Abbas does not have a frequent cleaning schedule. In fact, less than half of Bandar Abbas city's streets are cleaned on a regular basis usually once-a-week cleaning schedule. Therefore, a frequent cleaning schedule should be performed for streets with high pollutant loadings, especially in high traffic and industrial areas. Therefore, these results are important for the development of proper management strategies to decrease and prevent re-entrainment of ambient particulate matter by various sweeping technologies and ways.

Conclusions

From this work it can be concluded that there is significant contamination of street dust in Bandar Abbas, which has heavy vehicular traffic. Street dust is severely affected by traffic-related contaminants, including direct emissions from exhaust and wearing of brake pads and tires. More specifically, these direct emissions lead to high levels of PTEs such as Sb, Pb, Cu, Zn, and Cd, as well as four and three rings PAHs (e.g., Pyr, Phe, and Flu). The chemical signature of street dust is a good indicator of PTEs and PAHs accumulation in the urban surface environment. Since street dust is prevalent in many continental regions owing to resuspension by wind and anthropogenic activity (e.g., vehicles, construction), health risk calculations (according to land use and related models) are critical to determine health impacts and to motivate strategies to mitigate deleterious effects. The sum of the ILCRs through two exposure pathways, which was higher than 10^{-6} for adults and lower than 10^{-6} for children. Generally, children are exposed to more street dust pollutions than adults via each of the three pathways. The results of our study indicate that there is a need for additional research to find ways to reduce human exposure through interventions such as street cleaning and thereby reduce health risks from inhalation of street dust.

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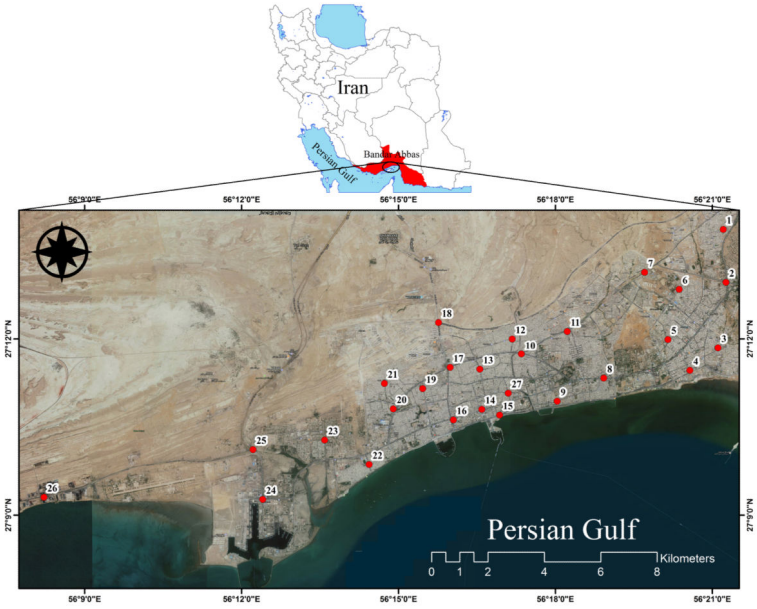

Fig. 1.
Sampling stations in Bandar Abbas city

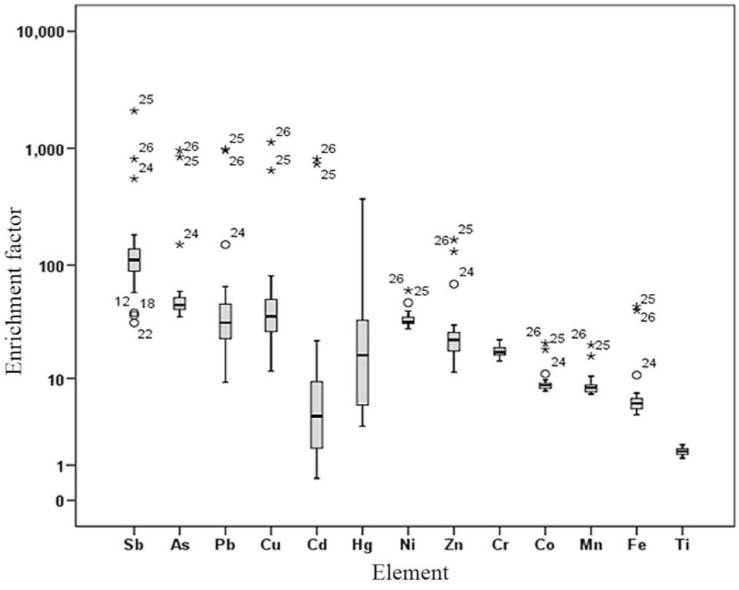

Fig. 2.

Box-plot of EF for PTEs in Bandar Abbas street dust. The numbers correspond to sample sites in Fig. 1

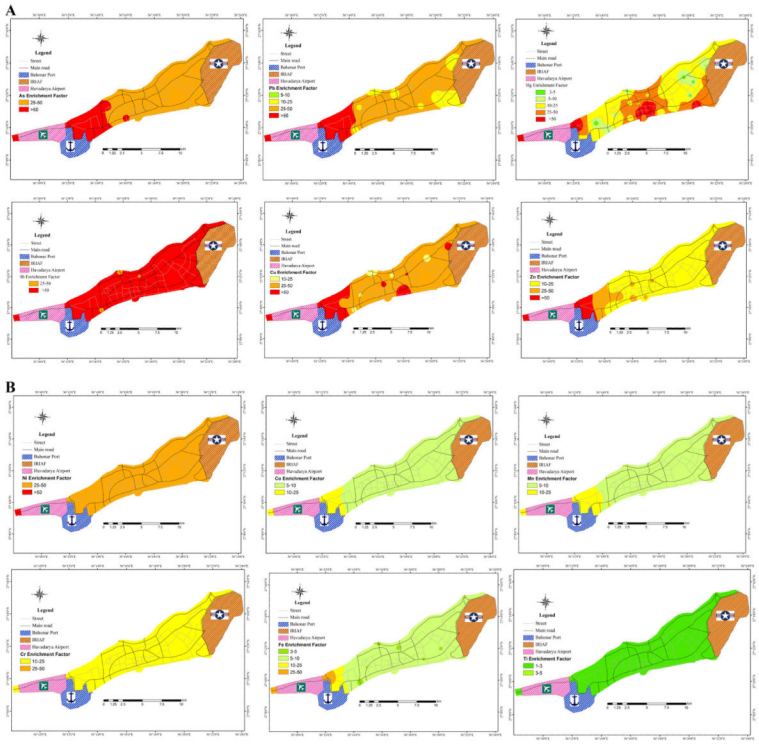

Fig. 3.
Spatial distribution of EF of PTEs in street dust of Bandar Abbas city

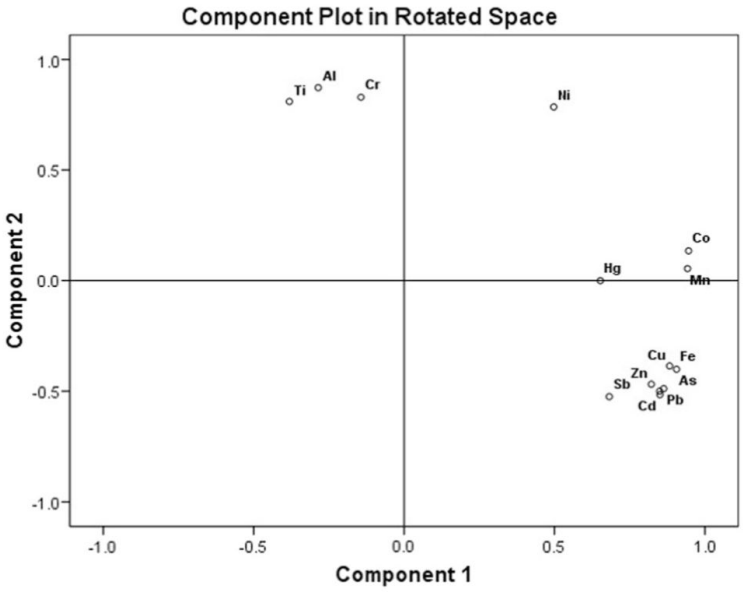

Fig. 4.
Principal component analysis (PCA) for PTEs

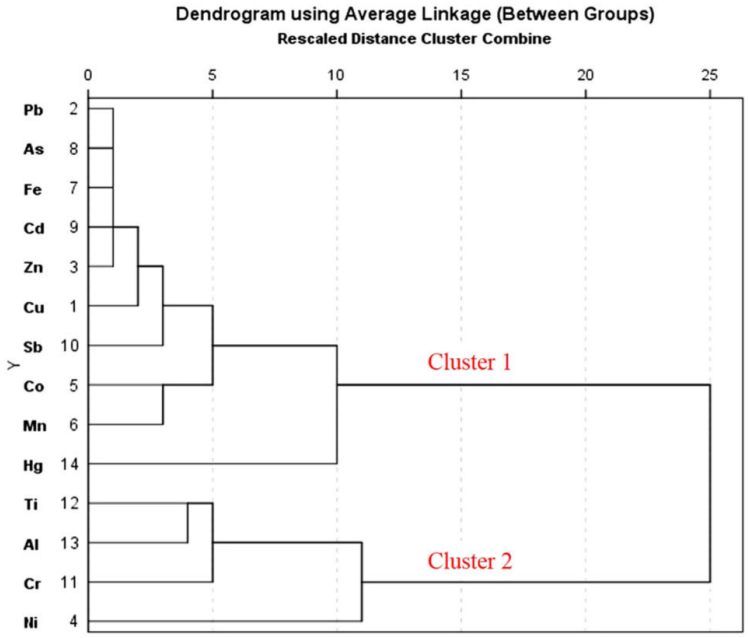

Fig. 5.
Cluster analysis (CA) for PTEs

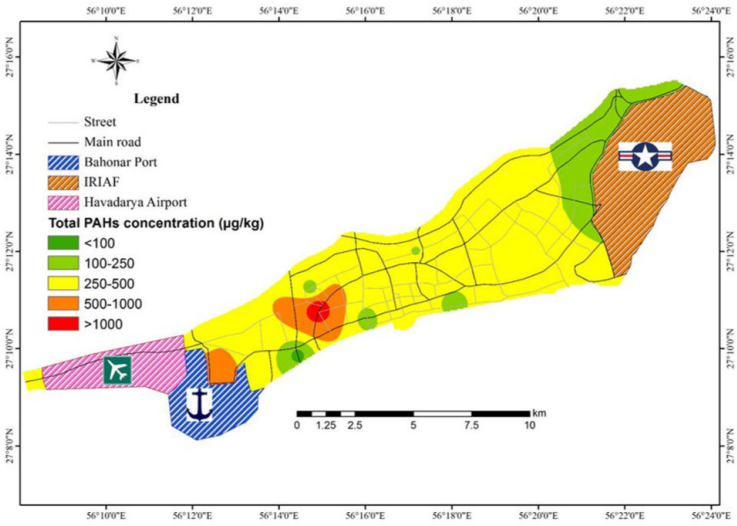

Fig. 6.
Spatial distribution of ΣPAHs in street dust of Bandar Abbas city

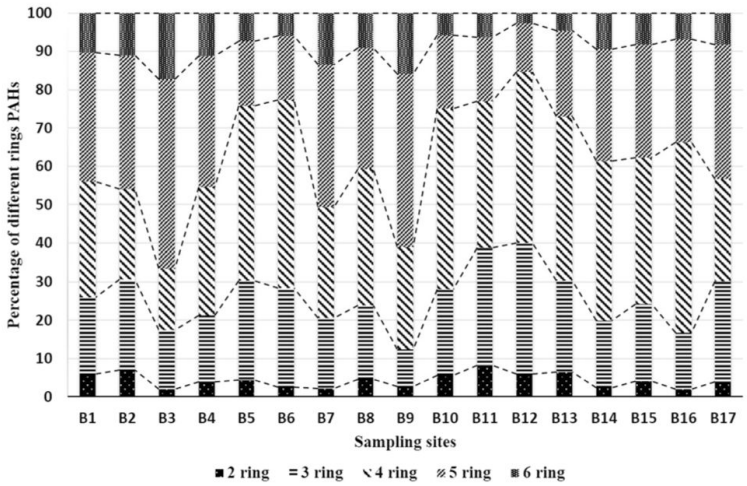

Fig. 7.

Percentage of PAHs with different numbers of aromatic rings in street dust of Bandar Abbas

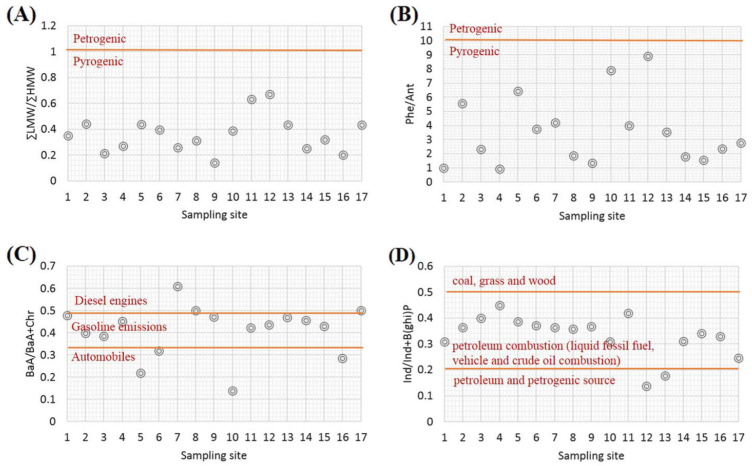

Fig. 8.

Source (petrogenic or pyrogenic) of PAHs in dust samples: **a** LMW/HMW; **b** phenanthrene/anthracene; **c** benzo(a)anthracene/(benzo(a) anthracene + chrysene); and **d** Indene/(Indene + Benzo[ghi] perylene)

Table 1

PTEs concentrations (mg kg^{-1}) in street dust collected in Bandar Abbas city

Elements	Mean	Median	Mode	K-S	S-W	VC	Std. Deviation	Variance	Skewness	Kurtosis	Minimum	Maximum	Upper Crust[b]	World Soil Average[a]
Cu	203.76	113.04	32.72	0.000	0.000	1.89	385.05	14,8261.9	3.80	14.61	32.72	1857.74	25	14
Pb	165.59	66.36	20.07	0.000	0.000	2	330.79	10,9423.3	3.36	10.31	20.07	1345.37	20	25
Zn	214.13	175.1	67.6	0.000	0.000	0.73	156.29	24,427.56	2.84	8.5	67.6	801.6	71	62
Ni	71	73.4	77.8	0.088	0.088	0.09	6.78	45.974	−0.53	−0.75	55.8	79.9	20	18
Co	9.58	9.2	9	0.089	0.011	0.14	1.37	1.866	1.32	3.06	7.1	13.8	10	6.9
Mn	542.48	535	565	0.000	0.000	0.12	66.94	4480.49	2.35	8.42	439	802	600	418
Fe	27,607.41	21,400	19,800	0.000	0.000	0.77	21,176.89	4.48	3.29	9.98	12,800	102,000	35,000	35,000
As	13.95	7.5	7.3	0.000	0.000	1.61	22.44	503.75	3.45	10.97	6.1	98.7	1.5	4.7
Cd	0.85	0.45	0.44	0.000	0.000	1.47	1.25	1.57	3.18	9.3	0.27	5.23	0.098	1.1
Sb	3.79	2.41	0.53	0.000	0.000	1.43	5.41	29.3	4.06	18.18	0.53	28.6	0.2	0.48
Cr	64.11	62.9	62.9	0.200	0.793	0.13	8.51	72.43	−0.22	−0.28	44.7	78.5	35	42
Ti	515.19	530	550	0.177	0.012	0.19	96.97	9402.85	−1.04	1.03	290	660	3000	3300
Al	8459.26	8800	9000	0.056	0.035	0.16	1364.74	1,862,507	−0.9	0.2	5300	10,300	80,400	71,000
Hg	0.2	0.11	0.02	0.000	0.000	1.5	0.3	0.09	2.72	7.2	0.01	1.22	0.05	0.1

K–S Kolmogorov–Smirnov test, *S–W* Shapiro-Wilt test, *VC* variation coefficients (VC = SD/mean)

[a] Kabata-Pendias and Pendias (2001)

[b] Rudnick and Gao (2003)

Table 2

Mean concentration of PTEs (mg kg^{-1}) in street dust in other countries and cities

City (Mean)	Population (Year)	Zn	Pb	Cd	As	Co	Cu	Cr	Hg	Ni	Sb	Reference
Bandar Abbas, Iran	680366 (2016)	214.13	165.59	0.85	13.95	9.58	203.76	64.11	0.2	71	3.79	This work
Bushehr, Iran	298945 (2016)	283.03	94.86	0.35	6.43	5.09	118	45.76	0.33	35.05	3.28	Abbasi et al., 2017
Tehran, Iran	13267637 (2016)	873.2	257.4	10.7	—	—	225.3	33.5	—	34.8	—	Saeedi et al., 2012
Ottawa, Canada	1323783 (2016)	184	68	0.6	2	11	188	59	0.06	19	—	Rasmussen et al., 2000
Accra, Ghana	2.27 million (2012)	327	120	—	—	—	110	124	—	—	—	Atiemo et al. 2012
Seoul, Korea	10.29 million (2016)	2840	120	—	—	—	65.8	43.3	—	—	—	Chon et al. 1995
Madrid, Spain	3.17 million (2016)	476	1927	—	—	3	188	61	–	44	—	De Miguel et al., 1997
Kavala, Greece	56371 (2014)	272	301	0.2	17	—	124	196	0.1	58	—	Christoforidis and Stamatis, 2009
Oslo, Norway	634293 (2014)	412	180	1.4	—	19	123	—	—	41	6	De Miguel et al., 1997
London, U.K.	8.79 million (2016)	372	370	2.7	—	—	80	—	—	—	—	Schwarz et al., 1988

Table 3

Correlation coefficients (*r*) of PTEs and ΣPAHs in Bandar Abbas street dust samples

PTEs	ΣPAHs	Cu	Pb	Zn	Ni	Co	Mn	Fe	As	Cd	Sb	Cr	Ti	Al	Hg	
ΣPAHs	1															
Cu	0.31	1														
Pb	**0.56** *	**0.55** **	1													
Zn	**0.75** **	**0.56** **	**0.78** **	1												
Ni	0.19	−0.04	0.05	0.32	1											
Co	**0.57** *	0.15	0.25	0.41*	**0.74** **	1										
Mn	0.19	0.36	0.456*	**0.57** **	**0.66** **	**0.68** **	1									
Fe	**0.54** *	**0.65** **	**0.76** **	**0.83** **	0.32	**0.56** **	**0.60** **	1								
As	0.41	**0.57** **	**0.75** **	**0.82** **	0.31	0.41*	**0.63** **	**0.84** **	1							
Cd	**0.69** **	**0.49** **	**0.77** **	**0.90** **	0.34	0.38	**0.52** **	**0.69** **	**0.78** **	1						
Sb	**0.77** **	**0.55** **	**0.68** **	**0.81** **	0.13	0.37	0.32	**0.74** **	**0.66** **	**0.65** **	1					
Cr	−0.31	0.11	0.06	0.13	0.478*	0.3	0.37	0.31	0.23	0.09	0.08	1				
Ti	−0.32	−0.12	−0.13	−0.17	0.36	0.29	0.13	0.16	−0.13	−0.28	−0.09	**0.70** **	1			
Al	0.07	−0.22	−0.21	0.05	**0.61** **	0.43*	0.2	0.08	−0.03	0.01	0.08	**0.71** **	**0.63** **	1		
Hg	0.19	0.45*	0.29	**0.52** **	0.41*	0.46*	0.34	**0.53** **	0.43*	0.43*	0.40*	0.19	0.01	0.12	1	
Compounds	Np	Ace	Fl	Phe	An	Flu	Pyr	B[a]a	Chr	B[e]p	B[b]f	B[k]f	B[a]p	Db[ah]a	B[ghi]p	Ind
Np	1	**0.750** **	**0.718** **	**0.762** **	**0.694** **	0.24	**0.569** *	**0.678** **	**0.771** **	0.43	0.418*	0.38	0.38	0.44	0.496*	0.48
Ace		1	**0.893** **	**0.824** **	**0.692** **	0.39	**0.510** *	**0.705** **	**0.730** **	0.42	0.382*	0.34	0.31	0.42	0.486*	0.41
Fl			1	**0.811** **	**0.615** **	0.38	0.46	**0.541** *	**0.703** **	0.44	0.419*	0.31	0.24	0.37	0.44	0.41
Phe				1	**0.601** *	**0.547** *	**0.553** *	**0.580** *	**0.733** **	0.30	0.448*	0.36	0.439*	0.45	0.494*	0.46
An					1	0.21	**0.523** *	**0.940** **	**0.884** **	0.169**	0.423**	0.368**	0.480*	0.253**	0.495**	0.177**
Flu						1	0.36	0.07	0.31	−0.03	0.26	0.13	0.21	0.24	0.21	0.21
Pyr							1	**0.508** *	**0.648** **	0.28	0.48	0.28	0.29	0.34	0.37	0.36
B[a]a								1	**0.817** **	0.497**	0.438**	0.374**	0.361*	0.472**	0.331**	0.417**
Chr									1	0.384**	0.142**	0.369**	0.418*	0.425**	0.382**	0.271**

PTEs	ΣPAHs	Cu	Pb	Zn	Ni	Co	Mn	Fe	As	Cd	Sb	Cr	Ti	Al	Hg	
B[e]p										1	**0.812** **	**0.613** **	**0.518** *	**0.670** **	**0.780** **	**0.726** **
B[b]f											1	**0.799** **	**0.598** *	**0.791** **	**0.864** **	**0.817** **
B[k]f												1	**0.696** **	**0.933** **	**0.923** **	**0.891** **
B[a]p													1	**0.737** **	**0.754** **	**0.710** **
Db[ah]a														1	**0.982** **	**0.960** **
B[ghi]p															1	**0.968** **
Ind																1

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

**Correlation is significant at the 0.01 level (2-tailed). Bold numbers indicate correlations >0.5

Table 4

Risk assessment of PTEs in street dust of Bandar Abbas city

					Adults				Children		
Element	**Oral RfD**	**Der RfD**	**Oral SF**	**Der SF**	**Hqing**	**HQder**	**HI = ΣHQi**	**Hqing**	**HQdermal**	**HI = ΣHQi**	**Cancer Risk**
Cu	4.00E-02	1.20E-02			1.18E-04	5.91E-04	7.10E-04	1.11E-03	5.91E-04	1.70E-03	
Pb	3.50E-03	5.25E-04			9.62E-05	4.81E-04	5.77E-04	8.98E-04	3.44E-03	4.34E-03	
Zn	3.00E-01	6.00E-02			1.24E-04	6.21E-04	7.46E-04	1.16E-03	4.45E-03	5.61E-03	
Ni- NC	2.00E-02	5.40E-03			8.25E-05	2.06E-04	2.89E-04	7.70E-04	1.48E-03	2.25E-03	
Ni- C											1.51E-09
Co- NC	2.00E-02	1.60E-02			1.11E-05	2.08E-05	3.20E-05	1.04E-04	1.49E-04	2.53E-04	
Co- C											2.56E-09
Mn	4.60E-02	1.84E-03			2.74E-04	1.03E-02	1.05E-02	2.56E-03	7.36E-02	7.61E-02	
As- NC	3.00E-04	1.23E-04			1.08E-03	3.95E-03	5.03E-03	1.01E-02	2.83E-02	3.84E-02	
As- C			1.50E + 00	3.66E + 00	1.24E-08	2.86E-09					2.78E-08
Cd- NC	1.00E-03	1.00E-05			1.98E-05	2.97E-03	2.99E-03	1.85E-04	2.13E-02	2.15E-02	
Cd- C											1.12E-10
Sb	4.00E-04	8.00E-06			2.20E-04	1.65E-02	1.67E-02	2.06E-03	1.18E-01	1.20E-01	
Fe	8.40E + 00	7.00E-02			7.64E-05	1.37E-02	1.38E-02	7.13E-04	9.84E-02	9.91E-02	
Cr- NC	3.00E-03	6.00E-05			4.97E-04	3.72E-02	3.77E-02	4.64E-03	2.67E-01	2.71E-01	
Cr- C											9.85E-10
Al	1.00E + 00	1.00E-01			1.97E-04	2.95E-05	2.26E-04	1.84E-03	2.11E-04	2.05E-03	
Hg	3.00E-04	2.10E-05			1.54E-05	3.29E-04	3.45E-04	1.44E-04	2.36E-03	2.50E-03	

Ing ingestion *Der* dermal contact, *C* carcinogenic, *Non C* non carcinogenic, *NC* non Cancer

Table 5

PAH concentrations (mg kg^{-1}) in street dust collected in Bandar Abbas city

Compounds	Aromatic rings	TEF	Mean	Median	Std. Deviation	Skewness	Minimum	Maximum
Naphthalene (Np)	2	0.001	16.28	10	24.38	4	4.8	110
Acenaphthene (Ace)	3	0.001	5.17	3.8	7.14	3.7	1.35	32
Fluorine (Fl)	3	0.001	7.91	4.7	11.83	3.61	0.53	52
Phenanthrene (Phe)	3	0.001	48.65	40	56.19	3.71	12	260
Anthracene (Ant)	3	0.01	17.53	15	15.25	2.05	3.4	65
Fluoranthene (Flu)	4	0.001	35.74	28	25.55	0.98	3.7	87
Pyrene (Pyr)	4	0.001	49.18	34	51.57	1.98	3.7	200
Benzo[a]anthracene (B[a]a)	4	0.1	19.8	15	22.12	2.78	2.1	95
chrysene (Chr)	4	0.01	27.71	15	29.98	2.8	6.2	130
Benzo[b]fluoranthene (B[b]f)	5	0.01	11.07	7.8	9.29	1.38	1.5	37
Benzo[k]fluoranthene (B[k]f)	5	0.01	22.04	14	17.22	1.05	3.4	63
Benzo[a]pyrene (B[a]p)	5	1	21.12	17	18.66	1.02	0.38	64
Benzo[e]pyrene (B[e]p)	5	1	25.32	23	17.31	0.97	6.2	66
Dibenzo[a,h]anthracene (Db [ah]a)	5	1	21.82	15	15.04	0.78	3.1	49
Benzo[ghi]perylene (B[ghi]p)	6	0.01	20.98	16	14.33	0.68	2.8	50
Indene(Ind)	6	0.1	12.04	7.8	9.88	1.06	0.6	36
ΣPAH			362.33	326.5	288.07	2.64	73.08	1339
2 ring			4.34	4.06	1.97	0.38	1.93	8.22
3 ring			21.46	20.06	6.02	0.24	9.72	34.02
4 ring			36.25	38.11	9.84	−0.42	15.83	49.37
5 and 6 ring			37.94	38.56	14.07	0.38	15.41	66.66
LMW			95.53	67.7	112.16	3.76	22.08	519
HMW			266.8	248.4	186.16	1.72	51	820
ComPAHs			244.98	227.6	174.55	1.82	47.9	772
ComPAHs/ΣPAHs			0.68	0.68	0.06	−0.23	0.57	0.78
CANPAHs/ΣPAHs			0.48	0.52	0.11	−0.12	0.3	0.68
NCANPAHs/ΣPAHs			0.52	0.48	0.11	0.12	0.32	0.7
TEQ			72.61	60.20	47.05	0.52	12.22	160.08
TEQ/ΣPAHs			0.21	0.23	0.7	0.04	0.1	0.33

Σ*PAHs* total PAH concentration, sum of individual mass concentration of 16 PAH congeners, *LMW PAHs* low molecular weight 2–3 ring PAHs, *HMW PAHs* high molecular weight 4–5ring PAHs, *COMPAHs* combustion derived PAH concentration, *CANPAHS* carcinogenetic PAHs, *NCANPAHs* non-carcinogenetic PAHs, *TEF* PAHs toxic equivalency factor with respect to BaP (Delistraty 1997; Sprovieri et al. 2007)

[a]Multiple modes exist. The smallest value is shown

Table 6

The results of principal component analysis (PCA) for PAHs in street dust Bandar Abbas city

	Component		
	Rotated Component Matrix		
	PC1	PC2	PC3
Np	0.26	**0.84**	0.02
Ace	0.19	**0.90**	0.12
Fl	0.15	**0.88**	0.13
Phe	0.26	**0.80**	0.43
An	0.60	**0.66**	−0.18
Flu	0.09	0.32	**0.83**
Pyr	0.20	**0.64**	0.21
B[a]a	0.61	**0.66**	−0.27
Chr	0.52	**0.77**	0.01
B[e]p	**0.71**	0.40	−0.42
B[b]f	**0.78**	0.50	−0.07
B[k]f	**0.93**	0.14	0.04
B[a]p	**0.77**	0.18	0.25
Db[ah]a	**0.94**	0.20	0.14
B[ghi]p	**0.94**	0.30	0.04
Ind	**0.93**	0.25	0.07

Extraction Method: Principal Component Analysis. Rotation Method Varimax with Kaiser Normalization. Principal factors ≥ 0.6 are highlighted in boldface

[a]Rotation converged in 6 iterations.

Table 7

Incremental lifetime cancer risk (ILCR) for children and adults exposed to Bandar Abbas street dust

	Children			Adults		
	ILCRsing	**ILCRsder**	**Cancer risk**	**ILCRsing**	**ILCRsder**	**Cancer risk**
Mean	1.43E-07	2.14E-05	2.16E-05	1.53E-08	2.99E-06	3.00E-06
Max	3.16E-07	4.72E-05	4.75E-05	3.38E-08	6.59E-06	6.62E-06
Min	2.41E-08	3.60E-06	3.63E-06	2.58E-09	5.03E-07	5.06E-07