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ORIGINAL PAPER

An assessment of heavy metals in dust at recreational parks in Trinidad, West Indies: contamination status, source identification and health risk implications

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Abstract

This study examined the contamination levels, origin specifications and potential health risk assessments of selected heavy metals (Cd, Cr, Cu, Mn, Ni, Pb and Zn) in road dusts from recreational parks throughout the island of Trinidad. Heavy metal concentrations were determined by Flame Atomic Absorption Spectroscopy (FAAS) and ranged from 0.33–2.30 µg/g, 2.15–23.33 µg/g, 12.73–103.12 µg/g, 46.37–419.37 µg/g, 8.61–30.00 µg/g, 13.38–323.87 µg/g and 78.20–669.19 µg/g for Cd, Cr, Cu, Mn, Ni, Pb and Zn, respectively. Indices such as geoaccumulation index, contamination factor and enrichment factor, applied to explore pollution patterns, indicated several degrees of pollution with the most prevalent polluting metals being Pb, Cu and Zn. Hazard index values were < 1 for all metals in the road dust indicating no significant risk from non-carcinogenic effect, while there were no significant cancer risks ($< 10^{-6}$) for both children and adults.

Keywords Toxic metals · Road dust · Recreational parks · Ecological risk · Health risk assessment · Trinidad

Introduction

Heavy metals are pervasive, non-degradable contaminants found throughout the environment in a variety of matrices including soil (Škrbić and Đurišić-Mladenović 2012), dust (Škrbić et al. 2012), sediment (Mohammed et al. 2017a, b), water (Mohammed et al. 2017a, b), cultivars (Škrbić and Čupić 2005), food commodities (Alves et al. 2017), baby foods (Škrbić et al. 2016), and functional products (Stilinović et al. 2014). The concentrations of these potential toxic elements in the environment have increased significantly since the inception of industrial activities across the globe. While some of these elements (Cu, Mn and Zn) are considered essential for biological and physiological functions, exposure to high levels could be potentially detrimental to the health of some organisms. Exposure to low levels of trace elements such Cd, Cr, Ni and Pb, can also have potentially negative health impacts on plants, animals and humans (Škrbić et al. 2020). The ecological and public health concerns about prolonged exposure to elevated levels of these elements arise because of their toxicity and carcinogenicity. The toxicity of heavy metals relates to their persistence in the atmosphere, bioavailability, and bioaccumulation in organisms, while its carcinogenicity is associated with epidemiological and experimental studies on the incidence of cancer risks based on exposure (Han et al. 2014; Shi et al. 2008).

The negative health effects linked to exposure to heavy metals may vary from headaches, insomnia, dizziness and joint pain to more serious ailments including respiratory disorders, cardiovascular diseases, risk of various types of cancers, disruption of the central nervous system, immune system deficiencies and breakdown of the endocrine system (Maina et al. 2018; Zhaoyong et al. 2018). For instance, chronic exposure to Cd can result in lung cancer, kidney failure and skeletal damage. Common health effects associated with Cr exposure are respiratory tract problems, stomach and small intestine issues, damage to the male reproductive system and cancer. Long-term exposure to Cu may result in irritation of the eyes, mouth and nose, nausea, and liver and kidney damage. Effects on the nervous system, irritation of the lungs and respiratory illnesses are related to high levels of exposure to Mn. Exposure to Ni can induce serious health

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effects include chronic bronchitis, diminished lung function and lung and nasal sinus cancers. Exposure to Pb can result in acute symptoms of headache, stomach pain, irritability, and restlessness. Long-term exposure in children can lead to neurological effects and decreased mental capacity with disturbances in their behavioural patterns and learning abilities. For adults, long-term effects can include kidney damage, decreased neurological functions, cardiovascular diseases, respiratory ailments and musculoskeletal effects and cancer. Short-term exposure to elevated concentrations of Zn can result in nausea, while introduction of Zn into the body over a prolonged period is associated with anemia and injury to the pancreas (Li et al. 2018; Moradi et al. 2020).

The ubiquitous nature of heavy metals, therefore, results in potential exposure to a wide cross-section of a population across various types of land uses and environments. Recreational parks are places where individuals and/or groups can spend their free time involved in relaxation activities, play, exercise, or socializing. Many recreational parks have evolved to accommodate the growing health trends across the globe, providing adequate space, equipment and facilities for the health-conscious individuals, persons wishing to unwind after work or those simply in search of a peaceful location. With the rapid increase in urbanization, and the need to preserve a sanctuary for relaxation, several recreational parks are in close proximities to developed areas, which could potentially diminish their environmental quality. At recreational parks, humans can be exposed to heavy metals through interaction with several media including soil, water, grass and dust, with the latter considered as the most intrusive (Al-Shidi et al. 2021; Chenery et al. 2020).

Dust can act as a major sink and source of many contaminants, including heavy metals, and is considered an important indicator for characterizing the quality of the surrounding environment and assessing health risks (Idris et al. 2020; Živančev et al. 2019). The ease of mobility of dusts allows for the possibility of wide range transport and deposition, and their large surface area promotes absorption and accumulation of metallic pollutants, resulting in a distribution of toxic elements that may have originated from remote locations. Local anthropogenic sources of heavy metals in dust include vehicular exhaust emissions, as well as non-exhaust such as wear and tear from tyres and brake lining, lubricating oil residues and degradation of paved surfaces (Casotti Rienda et al. 2023). In addition, the natural decomposition of plants and leaves, aided by weathering and runoff processes, may contribute to the presence of elevated levels within that matrix. Industrial activities, residential emissions and agricultural discharges can also be included as potential sources, as metal-laden dust particles can be easily re-suspended into the environment, traverse far distances, and accumulate via atmospheric deposition (Othman and Latif 2020; Zheng et al. 2010). The spatial distribution of heavy metals in dust has been found to be correlated with the population density and land use category, with increased concentrations at commercialized and high traffic areas as compared to rural areas (Duong and Lee 2011).

Dust particles, containing toxic heavy metals, can easily be re-suspended into the atmosphere, subsequently resulting in a potential health risk to individuals via three main exposure pathways: ingestion, inhalation, and dermal contact (Pragg and Mohammed 2020). Furthermore, a high proportion of heavy metals in dust can be absorbed into the human body (Jiang et al. 2018; Soltani et al. 2015; Song et al. 2018). Children are especially vulnerable owing to their low tolerance to pollutants, with ingestion and dermal contact as common exposure routes due to the regularity of hand to mouth activities. The high exposure of children is also due to the distance from the ground to their respiratory inlets, much closer to the level where dust is resuspended and mobilised from road surface. Numerous studies have reported on heavy metals in roadside dust, particularly in industrialized areas, with respect to distribution patterns, contamination assessments, source apportionment and the associated health risks to humans (Maina et al. 2018; Masto et al. 2019; Shabbaj et al. 2018; Xu et al. 2018). There have been several studies investigating the heavy metal content in soils at public parks and playgrounds, however, there is limited research on these toxic elements in dust (De Miguel et al. 2006; Glorennec et al. 2012; Han et al. 2017; Jin et al. 2019; Kasimov et al. 2019; Massas et al. 2010; Ng et al. 2003; Škrbić and Čupić 2004; Vlasov et al. 2021).

To obtain a comprehensive assessment of the status of heavy metal contamination in dust, appropriate indices may be utilized. Geo-accumulation index (I_{geo}), Enrichment factor (EF), Pollution Index (PI) and Potential Ecological Risk (E_r) are common pollution indices that provide a robust evaluation of metal-contaminated dust due to different geochemical backgrounds. Multivariate statistical analyses such as Principal Component Analysis (PCA) and Cluster Analysis (CA) have been reported as successful approaches for inferring sources of heavy metals in different matrices (Pan et al. 2017; Škrbić et al. 2005, 2010). These indices and statistical analyses can ascertain whether the accumulation of heavy metals may be attributed to natural or anthropogenic activities and are particularly useful for identifying specific sources and defining the potential risks due to accumulation (Ghanavati et al. 2018; Kowalska et al. 2018; Mazurek et al. 2019). Health risk assessments are useful tools for evaluating the toxicity of heavy metals based on the routes of exposure and have been used extensively for environmental media such as soils and road dust (Škrbić and Miljević 2002; Ferreira-Baptista and De-Miguel 2005; Gope et al. 2018; Škrbić et al. 2022).

The objectives of this present study are: (1) to ascertain the concentration of heavy metals in dust from various recreational parks throughout Trinidad, (2) to assess the contamination using different geochemical indices, (3) to assess the source of heavy metals in recreational park dust using multivariate statistical techniques and, (4) to determine the potential risk of exposure due to the presence of heavy metals in recreational park dust based on exposure pathways.

Materials and methods

Study area

The island of Trinidad is situated to the Northeast of Venezuela and Northwest of Guyana. Trinidad encompasses an area of 4800 km^2 and is the larger of the twin island nation of Trinidad and Tobago. The island experiences two climatic seasons—a dry period extending from December to May and a rainy season from June to November, and temperatures typically ranging between 25 and 29 °C. Within the island, there are many recreational parks that have been recently upgraded, based on the growing demand for accessible and safe facilities, that are conducive for relaxation, fitness, family gatherings and sporting events. Improvements have included adequate lighting, various types of exercise equipment and paved footpaths around the perimeter of the parks, ideal for walking, jogging or cycling. The four (4) recreational parks of interest for this study were selected based on popularity and geographical distribution.

The largest recreational park on the island is the Queen's Park (QP) Savannah (10.6687° N, 61.5144° W) and is considered one of the oldest recreational grounds in the West Indies. Activities within this park include annual kite flying competitions, numerous sporting activities, vending of the nation's cultural foods, and, particularly during the popular Carnival celebrations, several musical concerts and exhibitions. The Eddie Hart (EH) Savannah (10.6425° N, 61.3665° W) is located on the north-eastern region of the island and considered the second largest recreational park. Many recreational and leisure activities also occur within this locality, similar to the QP savannah, including the sale of popular street foods during the nighttime. Lange Park (LP) Savannah (10.5200° N, 61.4006° W) is centrally located within the island, providing recreational amenities, such as gym equipment and a children's playpark, to a main residential community. Skinner's Park (SP) Savannah (10.2671° N, 61.4621° W), situated in the southwestern region of the island, is located within the second largest city in Trinidad, and accommodates routine recreational and entertainment activities throughout the year. The control site, Barrackpore (BP) park (10.1885° N, 61.3806° W), a small recreational venue positioned within a rural area in the southern district of Trinidad, was selected based on the limited contributions from urbanized processes.

Sample collection and analysis

A total of twenty-four (24) sampling sites were identified at the various recreational parks (QP, $n = 8$; EH, $n = 5$; SP, $n = 4$; LP, $n = 4$ and BP, $n = 3$) and dust samples were collected along the paved pathways at each park. At each site, a polyethylene brush and scoop were used to collect a 200 g composite sample of dust. Samples were air-dried for three (3) days and sieved to a particle size of < 63 μm. A portion of dried sample (1 g) was allowed to pre-digest overnight in 10 mL concentrated nitric acid, followed by a digestion procedure at 135 °C for 6 h. After digestion, samples were allowed to cool, filtered, and diluted to a final volume of 50 ml using deionized water. Determination of Cd, Cr, Cu, Mn, Ni, Pb and Zn was achieved using flame atomic absorption spectrophotometry using respective calibration standards (Pragg and Mohammed 2020). These elements were investigated based on prevalence in the environment and toxicological significance.

Quality control and statistical analysis

A rigorous cleaning procedure was implemented, as all glassware were initially washed thoroughly with soap, soaked in an acid-bath and rinsed meticulously with deionized water before use. Reagent blanks were prepared in a similar manner as the dust samples, and analyses were conducted in triplicate for all samples and reported as mean ± standard deviation. Method validation was achieved by applying the full analytical procedure to a certified reference material (SRM 1944—New York/New Jersey Waterway Sediment). Percentage recoveries, limits of detection (LOD) and limits of quantification (LOD) are reported in Table 1. Recoveries were within the acceptable 80–120% analytical range and LOD and LOQ were determined as $3\sigma_{blank}$ and $10\sigma_{blank}$, respectively.

Inter-element relationships were assessed using Pearson's correlation. Source apportionment studies were evaluated using a combination of multivariate statistical tools including principal component analysis (PCA) and hierarchical cluster analysis (HCA). PCA was carried out using Varimax rotation with Kaiser Normalization after extraction of the data. For HCA, output data was obtained using Ward's linkage with squared Euclidean distance. All statistical treatment of data was determined using SPSS version 16.0.

Table 1 Percentage recoveries, limits of detection and limits of quantification

Element	SRM 1944 Measured value (µg/g)	% Recovery Certified value (µg/g)	LOD (µg/g)	LOQ (µg/g)	
Cd	8.57	8.88	97.4	0.21	0.69
Cr	277.5	266	85.5	0.33	1.10
Cu	328.4	380	86.4	0.32	1.08
Mn	430.9	505	85.3	0.39	1.29
Ni	77.9	76.1	102.4	1.05	3.52
Pb	349.1	330	105.8	0.80	2.65
Zn	606.4	656	92.4	0.35	1.18

Contamination assessments

Various indices have been previously established to assess heavy metal accumulation in dust, and usually requires a geochemical background for evaluation. The geochemical background should reflect natural processes with limited influence from anthropogenic activities. Generally, a reference geochemical background, reported as a global average from literature, may be applied. For this study, a local geochemical background (BP) was considered more appropriate for reliable characterization.

Geo-accumulation index (Igeo)

The geo-accumulation index, originally introduced by Müller (1981), is frequently applied for assessing contamination due to heavy metals in matrices such as soils and road dust (Škrbić et al. 2018; Cunha-Lopes et al. 2022). The I_{geo} was evaluated using Eq. (1):

$$\mathrm{I_{geo}} = \log_2\left(\frac{\mathrm{C}_n}{\mathrm{KB}_n}\right) \tag{1}$$

where C_n is the concentration of metal n in the road dust, B_n is the background concentration of metal n and K is a constant (≈ 1.5), used to counteract background variations. Classification of I_{geo} values are as follows: ($I_{geo} \le 0$) = not contaminated, ($0 < I_{geo} \le 1$) = not contaminated to moderately contaminated, ($1 < I_{geo} \le 2$) = moderately contaminated, ($2 < I_{geo} \le 3$) = moderately to strongly contaminated, ($3 < I_{geo} \le 4$) = strongly contaminated, ($4 < I_{geo} \le 5$) = strongly to extremely contaminated and ($I_{geo} > 5$) extreme contamination.

Pollution index (PI) and integrated pollution index (IPI)

These pollution indices (PI and IPI) are generally used together to ascertain the degree of pollution in road dust (Kowalska et al. 2018; Mazurek et al 2019). The PI was defined by Eq. (2):

$$\mathrm{PI} = \frac{\mathrm{C}_n}{\mathrm{C}_{ref}} \tag{2}$$

where C_n is the element concentration and C_{ref} is the baseline value of the corresponding element.

The IPI for each sample analysed was defined as the average value of each element's PI for that sample and was evaluated as shown in Eq. (3):

$$\mathrm{IPI} = \frac{PI_1 + PI_2 + .. + PI_i + .. + PI_n}{n} \tag{3}$$

where PI_i denotes the pollution index of the metal i and n represents the number of elements investigated. The classifications for both indices are highlighted in Table 2.

Enrichment factor (EF)

Enrichment factor is a useful technique for delineating the origination of heavy metals in the environment by normalizing against a reference metal element. The general EF equation (Eq. 4) was presented as:

$$\mathrm{EF} = \frac{[\mathrm{C}_n/\mathrm{C}_{ref}]\mathrm{Sample}}{[\mathrm{C}_n/\mathrm{C}_{ref}]\mathrm{Background}} \tag{4}$$

C_n represents the heavy metal concentration, B_n represents the metal background value, C_{ref} is the value of the reference element and B_{ref} is the background concentration of the reference metal. Conservative elements such as Al,

Table 2 Classifications of pollution index and integrated pollution index

PI value	Classification	IPI value	Classification
≤ 1	Low	≤ 1	Low
1 < PI ≤ 3	Middle	1 < IPI ≤ 2	Middle
≥ 3	High	IPI > 2	High

Fe, Mn, Sr and Ti are usually selected as references. For this study, Fe was selected as the reference element. Previous studies have indicated that values between 0.05 and 1.5 suggest natural sources, while EF values higher than 1.5 indicate anthropogenic origins (Ghanavati et al. 2018). Generally, however, the degree of contamination can be characterized into five classifications: (1) EF < 2 (minimal); (2) 2 ≤ EF ≤ 5 (moderate); (3) 5 ≤ EF ≤ 20 (significant); (4) 20 ≤ EF ≤ 40 (very high), and (5) EF > 40 (extremely high).

Potential ecological risk index (H') and potential ecological risk factor (E^i_r)

The potential ecological risk index is a comprehensive contamination assessment that considers the toxicity levels and ecological sensitivity of heavy metals in the environment (Kowalska et al. 2018). The assessment equation (Eq. 5), primarily introduced by Hakanson (1980), was denoted as:

$$H\prime = \sum_{i=1}^{n} E_r^i = \sum_{i=1}^{n} T_r^i \times C_f^i = \sum_{i=1}^{n} T_r^i \times C_s^i / C_n^i \tag{5}$$

where, n is the number of heavy metals, E^i_r is the potential ecological risk factor, T^i_r is the toxicity response coefficient for each metal, C^i_f is the contamination factor, C^i_s is the measured value for each metal, and C^i_n is the background value of an individual metal. The toxic response factors (T^i_r) of Cd, Cr, Cu, Mn, Ni, Pb and Zn are 30, 2, 5, 1, 5, 5 and 1, respectively. Table 3 highlights the categorizations and associated degrees of ecological pollution for E^i_r and H'.

Human health risk assessment model

Established by the USEPA (1996), the health risk assessment (HRA) model is commonly used to estimate the potential human health risks to adults and children based on exposure to contaminants. For this study, the HRA was applied to ascertain the potential non-carcinogenic (all elements under investigation) and carcinogenic risks (Cd, Cr and Ni) associated with heavy metal intake for the two receptors (i.e., children and adults).

Table 3 Classifications of E^i_r and H' based on degrees of pollution

Potential ecological risk factor, E^i_r	E^i_r classification	H'	H' classification
$E^i_r < 40$	Low	$H' < 150$	Low
$40 \leq E^i_r < 80$	Moderate	$150 \leq H' < 300$	Moderate
$80 < E^i_r \leq 180$	High	$300 \leq H' < 600$	High
$160 \leq E^i_r < 320$	Very high	$H' \geq 600$	Very high
$E^i_r \geq 320$	Dangerous		

For the three (3) exposure pathways (ingestion, inhalation, and dermal contact), Eqs. (6)–(8) were used to evaluate the potential health risks:

$$\mathrm{ADD_{ing}} = \mathrm{C} \times \frac{\mathrm{IngR} \times \mathrm{EF} \times \mathrm{ED}}{\mathrm{BW} \times \mathrm{AT}} \times 10^{-6} \tag{6}$$

$$\mathrm{ADD_{inh}} = \mathrm{C} \times \frac{\mathrm{InhR} \times \mathrm{EF} \times \mathrm{ED}}{\mathrm{PEF} \times \mathrm{BW} \times \mathrm{AT}} \tag{7}$$

$$\mathrm{ADD_{dermal}} = \mathrm{C} \times \frac{\mathrm{SL} \times \mathrm{SA} \times \mathrm{ABS} \times \mathrm{EF} \times \mathrm{ED}}{\mathrm{BW} \times \mathrm{AT}} \times 10^{-6} \tag{8}$$

$$\mathrm{LADD_{inh}} = \frac{\mathrm{C} \times \mathrm{EF}}{\mathrm{AT} \times \mathrm{PEF}} \times \left(\frac{\mathrm{InhR_{child}} \times \mathrm{ED_{child}}}{\mathrm{BW_{child}}} + \frac{\mathrm{InhR_{adult}} \times \mathrm{ED_{adult}}}{\mathrm{BW_{adult}}} \right) \tag{9}$$

$$C_{95\%\mathrm{UCL}} = \overline{X} + t_{\mathrm{a,n-1}} \cdot \left(\frac{\mathrm{S}}{\sqrt{n}} \right) \tag{10}$$

The average daily dose (ADD), expressed in mg/kg/day, is the daily dose intake through ingestion (ADD_{ing}), inhalation (ADD_{inh}), and dermal contact (ADD_{dermal}). To assess the cancer risk, the lifetime average daily dose ($LADD_{inh}$) for the inhalation exposure pathway for the carcinogenic elements was determined. C is the average concentration (mg/kg) of each heavy metal. *IngR* and *InhR* are the ingestion and inhalation rates, respectively. The ingestion exposure rates of soil and dust for children and adults are 200 mg/day and 100 mg/day, respectively (USEPA 2001). The inhalation exposure rates are 7.6 m^3/day and 20 m^3/day for children and adults, respectively (Van den Berg 1995). *EF* is the exposure frequency (days/year) and is generally site specific (Ferrieria-Baptista and De Miguel 2005). The value of *EF* for this study for both children and adults are 180 days/year. The exposure duration (*ED*), as recommended by the USEPA (2001) is 6 years for children and 24 years for adults. *SA* is the exposed skin surface area and SL is the skin adherence factor. For children, the values of *SA* and *SL* are 2800 cm^2 and 0.2 mg/cm^2/h, respectively, and for adults, the *SA* and *SL* values are 5700 cm^2 and 0.7 mg/cm^2/h, respectively (USEPA 2001). *ABS* is the dermal absorption factor (0.001). PEF (m^3/kg) is the particle emission factor with a value of 1.36×10^9 (USEPA 2001). *BW* is the average body for children (15 kg) and adults (70 kg), and *AT* is the averaging time (for non-carcinogens, $AT = 365 \times ED$ days; for carcinogens, $AT = 365 \times 70 = 25{,}550$ days) (USDOE 2004). $C_{95\%UCL}$ is the 95% upper confidence limit (UCL) of the arithmetic mean and is considered as an estimate of the "reasonable maximum exposure" based on concentration and exposure factors (USEPA 1989, Zhang et al. 2019). The concentrations of most elements in the matrix investigated approximate log-normal distributions and the exposure-point concentration for the transformed data can be calculated using Eq. 10, where

$\overline{X}$ (mg/kg) is the arithmetic mean concentration, the t-value measures the size of the difference relative to the variation in the sample data, a is associated with the confidence coefficient, n is the sample size and S is the standard deviation.

The characterization of non-carcinogenic risk was assessed using the hazard quotient (*HQ*) for each heavy metal at each location, while for carcinogens, the cancer risk (*CR*) was applied. The potential *HQ* is calculated using Eq. (11):

$$\mathrm{HQ} = \mathrm{ADD}/\mathrm{RfD} \tag{11}$$

where *ADD* (mg/kg/day) is the calculated dose for the three (3) exposure pathways from Eqs. (6)–(8) and *RfD* is the corresponding reference dose, defined as the intake per unit of body weight (USEPA 1996). The hazard index (HI) is the sum of the HQ values for each exposure pathway as denoted by Eq. (12). HI values < 1 indicates no significant health risks due to non-carcinogenic effects, whereas HI values > 1 suggest adverse health effects might occur.

$$\mathrm{HI} = \mathrm{HQ}_{\mathrm{ing}} + \mathrm{HQ}_{\mathrm{inh}} + \mathrm{HQ}_{\mathrm{dermal}} \tag{12}$$

The cancer risk, *CR*, is considered as the probability of developing cancer, resulting from prolonged exposure to carcinogenic hazards (Ghanavati et al 2018; Zhang et al 2019). The following equation was used to ascertain the carcinogenic risks due to exposure to heavy metals during a lifetime (USEPA 1996):

$$\mathrm{CR} = \mathrm{LADD} \times \mathrm{CSF} \tag{13}$$

where *LADD* is the lifetime average daily dose, and is calculated for each carcinogenic element investigated for the inhalation route of exposure based on Eq. 14:

$$LADD = \frac{\mathrm{C} \times \mathrm{EF}}{\mathrm{AT}} \times \left(\frac{\mathrm{CR}_{\mathrm{child}} \times \mathrm{ED}_{\mathrm{child}}}{\mathrm{BW}_{\mathrm{child}}} + \frac{\mathrm{CR}_{\mathrm{adult}} \times \mathrm{ED}_{\mathrm{adult}}}{\mathrm{BW}_{\mathrm{adult}}} \right) \tag{14}$$

The cancer slope factor (*CSF*), expressed as per mg/kg/day, relates the elemental daily intake during lifetime exposure to the incremental risk of developing cancer (USEPA 1996). *CR* values > 10^{-4} are considered unacceptable and indicate a significant carcinogenic risk, whereas values within the range of 10^{-6} to 10^{-4} suggest a potential cancer risk. Values < 10^{-6} signify negligible risk and are not regarded as significant (Gope et al. 2018).

Results and discussion

Heavy metal concentrations in park dust

The heavy metal concentration in dust samples at the various recreational parks, in comparison to the background values and Canadian Council of Ministers of the Environment (CCME) guidelines for soils at residential/parkland areas (2007), are highlighted in Table 4. The variation of heavy metals in dust between the parks investigated is illustrated in Fig. 1. For all parks, the order of heavy metal concentrations decreased in the following order: Zn > Mn > Pb > Cu > Ni > Cr > Cd.

The concentrations of Cd in the dust samples across all parks ranged from 0.33 to 2.30 μg/g with an average concentration of 0.75 μg/g. With the exception of EH, average Cd concentrations at the parks were higher than the reported background value of 0.52 μg/g and the average Cd levels at all parks were below the CCME guidelines established for soils at residential/parkland areas. The absence of major industrial activities around the parks investigated in this study, suggests that contributions of Cd in the dust may be attributed to natural sources. Notwithstanding, there exists the possibility of contributions from vehicular sources, specifically degradation of tyres, vehicular and brake lining wear and tear, and deposition of lubricating oils on the surfaces of the roadways (Qiang et al. 2015; Alsbou and Al-Khashman 2018). In addition, the preservation of grass within parks involves frequent maintenance, where fertilizer treatments and the use of agricultural equipment are utilized. Different types of fertilizers (particularly phosphatic) contain varying concentrations of Cd, which can eventually accumulate in dust, after continuous applications (Alloway 2013). The elevated concentrations of Cd observed at SP and QP, as compared to the other parks, may originate from anthropogenic origins, resulting from the higher traffic densities that are typically observed at those particular locations.

For all parks, an average Cr concentration of 8.00 μg/g was recorded, with values ranging from 1.48 to 15.97 μg/g. This average was lower than the background value of 10.39 μg/g and the CCME guideline value of 64 μg/g. The average Cr concentration reported for SP, however, was minimally higher than the background value, when compared to EH, LP and QP. The presence of Cr in dust is usually attributable to parent materials contained within the dust when specific sources are non-existent (Qiang et al. 2015). Another common contributor to Cr in soils and road dust is the use of chemical treatments (traditionally used for the preservation of wooden light poles) containing Cu, Cr, and As (CCA) (Alloway 2013). Vehicular emission is also considered a potential source of Cr in dust at urbanized areas (Ferreira-Baptista & De Miguel 2005; Praveena & Aris 2018).

Copper levels in the dust samples from the four recreational parks ranged from 10.49 – 103.12 μg/g, with a mean value of 36.36 μg/g. The average Cu concentrations for each park were higher than the background value (10.48 μg/g), but lower than the established CCME guideline concentration (63 μg/g). Copper is considered an essential element which occurs naturally in its elemental form and is part of

Table 4 Heavy metal concentrations (μg/g) in recreational park dusts of trinidad compared with guideline values

Location		Cd	Cr	Cu	Mn	Ni	Pb	Zn
EH	Min	0.33	1.48	12.73	105.99	9.93	19.71	78.20
	Max	0.75	6.22	38.96	340.60	21.73	65.88	212.59
	Mean	0.49	2.95	21.22	204.61	15.11	33.62	129.87
	SD	0.18	1.93	10.61	109.57	5.08	18.57	51.66
SP	Min	0.97	9.36	30.56	49.34	24.79	43.98	236.23
	Max	1.30	13.86	61.17	288.79	29.94	87.00	358.66
	Mean	1.14	12.31	40.87	136.02	27.18	63.60	293.87
	SD	0.18	2.02	13.82	106.91	2.44	21.66	63.21
LP	Min	0.62	5.29	10.49	39.10	18.78	16.58	98.07
	Max	0.83	7.66	17.10	188.62	29.01	20.01	210.33
	Mean	0.73	6.68	14.55	112.40	23.13	18.21	156.18
	SD	0.09	1.09	2.84	68.58	4.95	1.41	53.42
QP	Min	0.33	4.25	33.45	111.67	8.61	49.31	112.30
	Max	2.30	15.97	103.12	339.01	28.44	401.25	669.19
	Mean	0.73	9.65	54.74	227.22	18.61	180.05	307.99
	SD	0.67	5.17	21.56	89.78	6.30	124.68	173.93
All parks	Min	0.33	1.48	10.49	39.10	8.61	16.58	78.20
($n = 21$)	Max	2.30	15.97	103.12	353.57	29.94	401.25	669.19
	Mean	0.75	8.00	36.46	182.59	20.27	92.18	233.97
	SD	0.46	4.74	22.51	99.90	6.52	103.85	137.76
Background[a]	Min	0.38	9.00	9.72	221.87	15.41	16.23	61.32
	Max	0.59	11.81	11.28	312.75	17.62	17.66	97.74
	Mean	0.52	10.39	10.48	268.84	16.60	17.03	73.60
	SD	0.12	1.40	0.78	45.52	1.11	0.73	20.91
CCME soil guidelines[b]		10	64	63	–	45	140	250

[a]Concentrations obtained from BP

[b]Residential/Parkland (Canadian Council Ministers of Environment 2007)

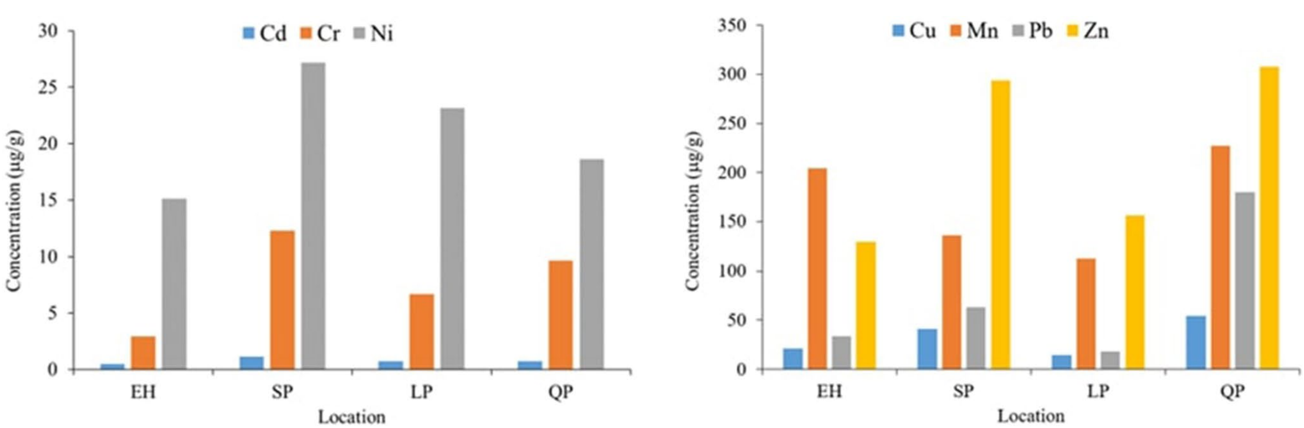

Fig. 1 Variation of heavy metals in dust between the four recreational parks in Trinidad for **a** Cd, Cr and Ni and **b** Cu, Mn, Pb and Zn

many mineral components, resulting in it being widely distributed within the environment. Anthropogenic sources of Cu, particularly in soils and dust from parkland locations, include deposition of dust and aerosol particles from vehicular emissions, degradation of automotive components (brake linings, bearings, bushings, etc.), micronutrient fertilizers, agricultural chemicals and wood preservatives (CCA) (Alloway 2013; Alsbou & Al-Khashman 2018; Jiang et al. 2018).

The mean value for Mn concentration in dust samples from the four parks was 182.59 μg/g (39.10 μg/g to 353.57 μg/g). The average Mn content at each recreational park was lower than the background concentration of

268.84 μg/g, indicating that this particular element originated mainly from natural sources. Typically, elevated levels of Mn in dust suggest contributions from anthropogenic sources and are usually attributed to industrial (steel works) and agricultural effluents (pesticides and fertilizers) (Casotti Rienda and Alves 2021).

An average Ni concentration of 20.27 μg/g (8.61–29.94 μg/g) was observed for samples obtained from the four recreational parks, where each park, except EH, reported an average Ni content higher than the background value (16.60 μg/g). All parks, however, recorded mean Ni levels below the recommended CCME guideline values (45 μg/g) for residential/parkland locations. Elevated levels of Ni in dust, particularly at urban areas, may be associated with degradation of brake lining and vehicular emissions (Du et al. 2013; Han et al. 2014; Praveena & Aris 2018).

The Pb content is the dust samples ranged between 16.58 and 401.25 μg/g for all parks, with an observed average concentration of 92.18 μg/g. The mean Pb concentration at the individual parks were higher than the background value of 17.03 μg/g, while all parks, except QP, reported Pb levels lower than the CCME guideline value (140 μg/g). Typically, high concentrations of Pb in road dust are associated with traffic burden, emissions from leaded gasoline and industrial activities. The phasing out of leaded gasoline occurred more than a decade ago within this nation, however, the ubiquitous nature of the element allows for residual Pb to still linger in the environment. Lead compounds are typically added to engine lubricant oils as a preventative measure for wear and tear (Mang and Dresel 2007; Qiang et al. 2015). The elevated levels of Pb at QP is not surprising, given that this location is within the capital city of Trinidad and the volume of traffic surrounding this park is much higher compared to the other parks of interest.

The Zn content for the four parks varied between 78.20 and 669.19 μg/g with an average concentration of 233.97 μg/g, which was higher than the baseline value (73.60 μg/g), but lower than 250 μg/g established by the CCME. For each park, the average Zn levels were higher than the control site, however, the mean Zn concentrations at SP and QP exceeded the CCME value established for parkland soils. The presence of Zn in road dust is usually attributed to traffic emissions, abrasion of vehicular parts and degradation of tyres and brake pads/liners as this element is used in the rubber vulcanizing process of tyre manufacturing and as an additive in lubricant oils (Mang and Dresel 2007; Qiang et al. 2015). The high Zn levels at SP and QP, compared to EH and LP, may be attributed to the differences in traffic densities, as SP and QP are within main towns whereas EH and LP are encompassed within residential locations.

While the heavy metal content in dust around recreational parks may be significantly attributed to traffic-related sources, the possibility of contributions from decomposed, degraded plant material cannot be ignored. The uptake of heavy metals from soils, which can translocate through the roots of trees, may eventually accumulate in leaves over time. In addition, atmospheric pollution, which may have originated from remote locations, may result in the deposition of metal-laden dust onto the surfaces of branches and leaves. Once deposited on the ground, disintegration of plant material can produce small particles, due to various processes, which can be integrated into a complex dust composition.

Table 5 provides a parallel assessment of heavy metal content at local recreational parks versus selected studies from across the world. Except for Hong Kong, Cd levels in EH, SP, LP and QP were comparable to those reported at various locations. Chromium concentrations in local park dust were typically lower than levels obtained for other regions of the world. The Cu content at all locations investigated for this study were similar to those reported previously in China (with the exception of Nanjing and Hong Kong), Poland, Thailand and Turkey, while Mn concentrations were lower than levels observed in Baotou, Beijing, Nanjing, Urumqi and Hong Kong. Similar Ni concentrations were observed for all locations, except Nanjing, while various Pb levels were reported for all studies. The Pb content at EH, SP and LP were comparable to those stated for Baotou, Changchun, Tianjin, Urumqi, Hong Kong, Serbia and Turkey, while the Pb concentration at QP was comparable to the study conducted in Beijing but higher than all other locations. The concentrations of Zn in dust at all four local parks were higher than those reported for Baotou, Urumqi, Thailand and Turkey (but lower than Nanjing and Hong Kong), and similar to levels observed in Beijing, Changchun and Poland. In general, similarly low levels that were observed between local parks and other regions, particularly for Cd and Ni, may be attributed to natural levels, while the variations for the other elements may be credited to specific, anthropogenic activities within those districts.

Contamination assessments of heavy metals in recreational park dust in Trinidad

Boxplots of I*geo* values obtained from the analysis of road dust for heavy metals at the four recreational parks are illustrated in Fig. 2. For EH (Fig. 2a), average I*geo* values were < 0 for Cd, Cr, Mn and Ni, signifying no contamination, while average values of I*geo* between 0 and 1 were observed for Cu, Pb and Zn, indicating no contamination to moderate contamination. Evaluation of average I*geo* values for SP (Fig. 2b) revealed no contamination by Cr and Mn (I*geo* < 0), minimal to moderate contamination by Cd and Ni, and moderate contamination ($1 <$ I*geo* < 2) by Cu, Pb and Zn. For all elements except Zn (Fig. 2c), no

Table 5 Comparison of heavy metal concentrations (μg/g) in recreational park dusts of Trinidad with other cities

Location, Country	Cd	Cr	Cu	Mn	Ni	Pb	Zn	References
Tacarigua, Trinidad (EH)	0.49	2.95	21.22	204.61	15.11	33.62	129.87	This study
San Fernando, Trinidad (SP)	1.14	12.31	40.87	136.02	27.18	63.60	293.87	This study
Chaguanas, Trinidad (LP)	0.73	6.68	14.55	112.40	23.13	18.21	156.18	This study
Port-of-Spain, Trinidad (QP)	0.73	9.65	54.74	227.22	18.61	180.05	307.99	This study
Baotou, China	0.3	154.1	26.9	504.4	25.1	36.2	49.7	Han et al. (2017)
Beijing, China	0.64	69.33	72.13	–	25.97	201.82	219.20	Du et al. (2013)
Beijing, China	–	78.87	52.06	521.7	22.25	80.27	–	Jin et al. (2019)
Changchun, China	0.33	59.28	37.82	–	23.08	69.12	169.26	Qiang et al. (2015)
Nanjing, China	1.92	133	141	602	115	119	585	Wang et al. (2016)
Tianjin, China	0.60	47.6	62.7	–	18.4	28.4	–	Živančev et al. 2019)
Urumqi, China	0.71	30.97	42.54	499.58	35.73	43.22	94.97	Zhaoyong et al. (2018)
Hong Kong	7.0	263	143	518	–	77.3	1883	Ng et al. (2003)
Krakow, Poland	0.80	16.3	55.5	–	10.5	120.2	176.7	Gąsiorek et al. (2017)
Maha Sarakham, Thailand	0.05	–	12.04	–	–	6.39	43.76	Ma and Singhirunnusorn (2012)
Novi Sad, Serbia	0.47	36.1	50.1	–	21.0	66.5	–	Škrbić et al. (2018)
Istanbul, Turkey	1.05	54.66	37.54	–	39.23	38.98	91.92	Demir et al. (2016)

significant contamination was identified at LP (I*geo* < 0), with minimal to moderate contamination being observed for Zn (1 < I*geo* < 2). At QP (Fig. 2d), no contamination by Cd, Cr, Mn and Ni was observed (I*geo* < 0), however there was moderate contamination by Cu and Zn (1 < I*geo* < 2) and moderate to strong contamination by Pb (2 < I*geo* < 3). It is evident that contamination at the recreational parks is attributed mainly to Cu, Pb and Zn, which are indicative of contributions from vehicular sources, with contrasting degrees of contamination across the parks credited to distinct land uses and traffic-related activities.

The range of PI values of each element and average IPI values for each recreational park are highlighted in Table 6. Most locations at EH were classified as low-level and middle-level (PI ≤ 3), apart from one sample point which was categorized as high-level (PI ≥ 3) for Cu and Pb. A similar trend was observed for all areas within LP, where PI values were typically ≤ 3, suggesting low-level and middle-level contamination. Comparable ranges were observed for SP and QP for Cd (except one location), Cr, Mn and Ni with PI values ≤ 3, indicating low-level to middle-level contamination. The majority of PI values at those two parks for Cu, Pb and Zn were ≥ 3, signalling high-level contamination. Based on average IPI values, both EH and LP were classed at middle-level contamination (1 < IPI ≤ 2), while both SP and QP were identified as parks with a high degree of contamination (IPI > 2).

Figure 3 displays the EF values for each metal at the various recreational parks. At EH, minimal enrichment occurs for Cd, Cr, Mn and Ni (EF < 2), with a moderate enrichment of Cu, Pb and Zn (2 ≤ EF ≤ 5). A similar trend is observed at SP, where minimal enrichment exists for Cr, Mn and Ni, and moderate enrichment for Cd, Cu, Pb and Zn. For all elements at recreational park LP, minimal enrichment occurs at all locations within the park, as denoted by their EF values being lower than 2. Within QP, minimal enrichment exists for Cd, Cr, Mn and Ni (EF < 2), with moderate enrichment for Zn (2 ≤ EF ≤ 5) and significant enrichment for Cu and Pb (5 ≤ EF ≤ 20). Based on the findings, it can be postulated that Cd, Cr, Mn and Ni may be considered to have originated from natural sources at the respective parks as signified by their low EF values.

The potential ecological risk assessment of heavy contamination for each park is displayed in Table 7. With the exception of Cd for SP, LP and QP and Pb for QP ($40 \leq E^i_r < 80$), the potential ecological risk factors, E^i_r, for all elements were < 40, suggesting low degrees of pollution with respect to those particular elements at the specified locations. While moderate degrees of pollution for Cd (at SP, LP and QP) and Pb (at QP) may be attributed specifically to vehicular contributions (wear and tear of brake lining and tyres, oil depositions), the influence of the various toxic response factors applied for each element must be considered. Nevertheless, the potential ecological risk index values, H', for all locations were < 150, indicating a low degree of contamination for all parks.

Health risk assessment of heavy metals in recreational park dust

The results of the health risk assessments based on exposure to heavy metals in dust at the various recreational parks are highlighted in Table 8. Based on the three main exposure pathways (ingestion, inhalation, and dermal contact), the

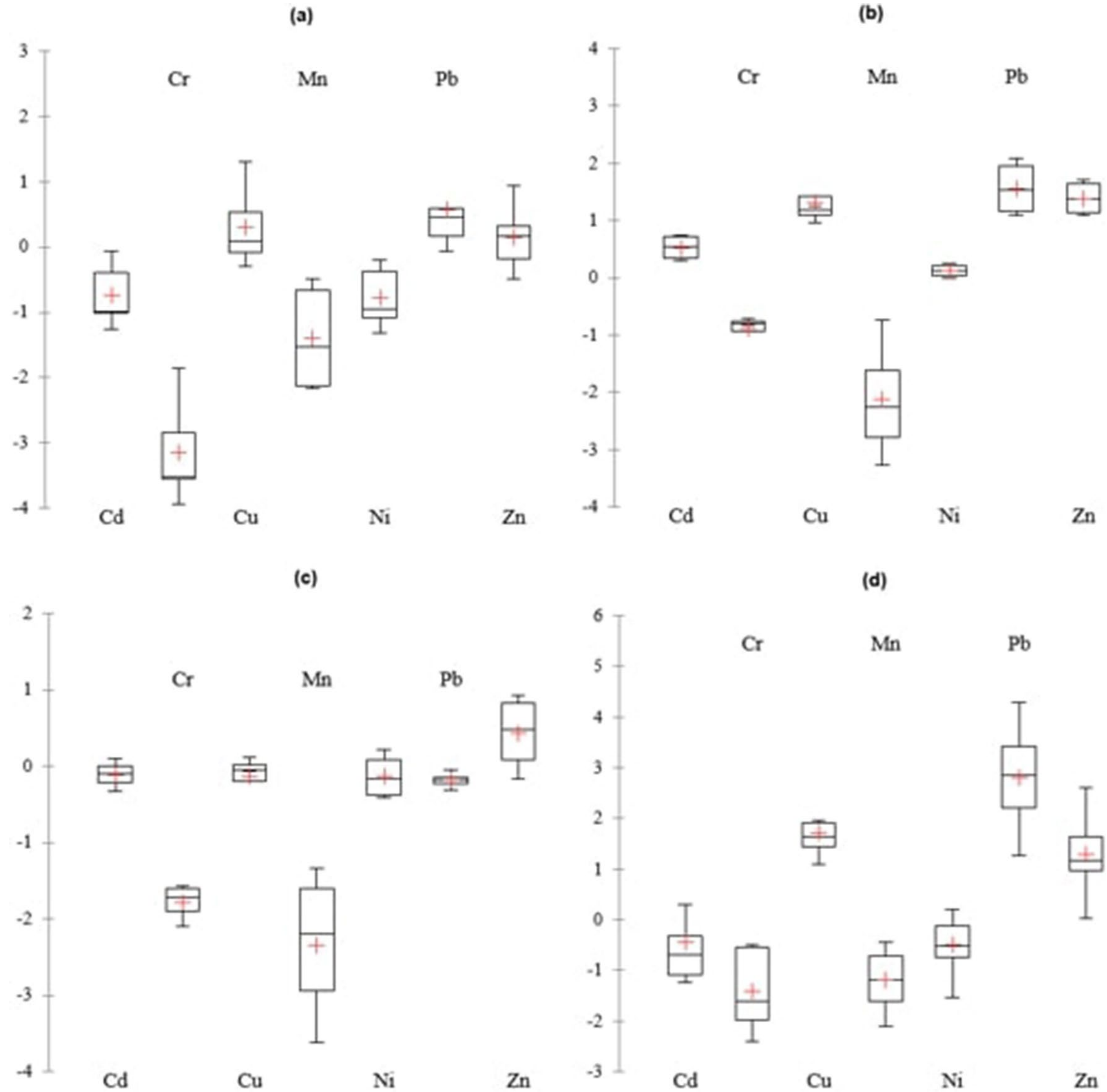

Fig. 2 I_{geo} values for heavy metals in dust at **a** EH, **b** SP, **c** LP and **d** QP

Table 6 PI range and average IPI values for each recreational park in Trinidad

Location	PI range							Average IPI
	Cd	Cr	Cu	Mn	Ni	Pb	Zn	
EH	0.63–1.44	0.14–0.60	1.21–3.72	0.39–1.27	0.60–1.31	1.16–3.87	1.06–2.89	1.24
SP	1.86–2.50	0.90–1.33	2.92–5.84	0.18–1.07	1.49–1.80	2.58–5.11	3.21–4.87	2.45
LP	1.20–1.60	0.51–0.74	1.00–1.63	0.15–0.70	1.13–1.75	0.97–1.18	1.33–2.86	1.20
QP	0.64–4.42	0.41–1.54	3.19–9.84	0.42–1.32	0.52–1.71	2.90–23.56	1.53–9.09	3.47

non-carcinogenic and carcinogenic risks were established. Evaluation of the HQ values for each element at the recreational parks revealed that the orders of exposure for both children and adults were ingestion > dermal contact > inhalation, which were consistent with previous studies (Ma and Singhirunnusorn 2012; Safiur Rahman et al. 2019; Pragg and Mohammed 2020; Kabir et al. 2021). For the four parks under investigation, HI values of all elements were < 1 for both children and adults, signifying negligible non-carcinogenic risks due to exposure. Of particular interest, the HI values at all parks were higher for children than for adults, suggesting a greater risk due to exposure to heavy metals

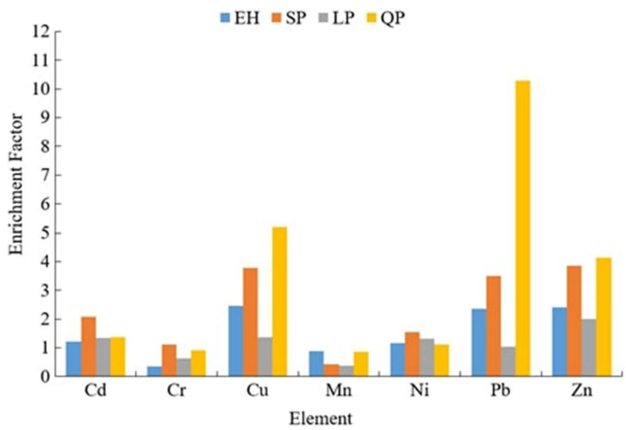

Fig. 3 Enrichment factor for each element at the four recreational parks

in dust for that age category. There is, generally, a higher susceptibility of risk for children due to lower body weights, reduced tolerance levels to pollutants and their behavioural patterns during outdoor activities (Han et al. 2017; Keshavarzi et al. 2018). The cancer risks for Cd, Cr and Ni based on inhalation were evaluated as $< 10^{-6}$, indicating that no potential carcinogenic risks exist due to exposure to those individual elements. Even though the carcinogenic risks are deemed acceptable based on the threshold values established by environmental and regulatory agencies, consideration must be given to the following limitations: (1) the carcinogenic risk assessments were based on the inhalation exposure pathway only, possibly resulting in an underestimation of the overall cancer risk; (2) the cancer risk value for Cr is based on total Cr concentrations, as opposed to Cr(VI) only, which is more toxic than Cr(III); and (3) the parameters used for evaluations are generalized by the US EPA and may differ from one particular region to the next (Mohammed and Mohammed 2022).

Correlation coefficient analysis

To identify inter-elemental links and evaluate potential sources of heavy metals in recreational park dust based on their correlation levels, correlation coefficient analyses were conducted, and the results displayed in Table 9. Pearson's correlation revealed significant positive relationships between Cr and Cu ($r = 0.684$, $p < 0.01$), Cr and Ni ($r = 0.725$, $p < 0.01$), Cr and Zn ($r = 0.779$, $p < 0.01$), Cu and Pb ($r = 0.877$, $p < 0.01$), Cu and Zn ($r = 0.832$, $p < 0.01$), Ni and Zn ($r = 0.601$, $p < 0.01$), and Pb and Zn ($r = 0.793$, $p < 0.01$). Based on the correlations observed, positive associations among Cd, Cr and Ni and among Cu, Pb and Zn suggested contributions mainly from a combination of vehicular-related sources. Degradation from tyres and brake pads, abrasions from car and wheel components, leakages from batteries and lubricating oils and exhaust emissions are all potential sources of these elements in dust (Praveena and Aris 2018; Sadeghdoust et al. 2020; Shabanda et al. 2021). The strong positive correlation between Cr and Cu may also be associated with the usage of the wood preservative containing those elements, that were used predominantly to treat wooden light poles, which are still found within some recreational parks today. In addition, the absence of any correlation between Mn and the other elements suggests the possible origination of that heavy metal from mainly natural sources.

Principal component and hierarchical cluster analyses

Multivariate statistical tools (PCA and HCA) have been used successfully for environmental assessments by reducing the information obtained into smaller, valuable factors, while preserving the validity of the dataset (Pan et al. 2017). Particularly for PCA, the original data must be validated using the Kaiser–Meyer–Olkin (KMO) measure of sampling adequacy (> 0.5), and Bartlett's test of sphericity ($p < 0.01$) and the findings of PCA are used in conjunction with HCA as a confirmatory tool in source apportionment studies. For this study, the established conditions for PCA were fulfilled, as indicated by the KMO value of 0.659 and the Bartlett's test of sphericity value < 0.01. Using the Varimax method and Kaiser Normalization, three rotated components with Eigenvalues > 1 were extracted, with PC1 accounting for 54.739% of the total variance, PC2 explaining 18.983% of the total variance and PC3 representing 14.586% of the total variance (Table 10). Component 1 was highly loaded with Cu (0.947), Pb (0.906) and Zn (0.903), component 2 was heavily influenced by Cd (0.890), Cr (0.720) and Ni (0.784) and component 3 was heavily loaded with Mn (0.965).

Table 7 Potential risk assessment of heavy metal contamination in various recreational park dusts in Trinidad

Location	Average potential ecological risk factor (E^i_r)							Average H'	Risk assessment
	Cd	Cr	Cu	Mn	Ni	Pb	Zn		
EH	28.34	0.57	10.12	0.76	4.55	9.87	1.76	55.98	Low
SP	65.56	2.37	19.50	0.51	8.19	18.67	3.99	118.79	Low
LP	41.99	1.29	6.94	0.42	6.97	5.35	2.12	65.07	Low
QP	42.03	1.86	26.11	0.85	5.61	52.86	4.18	133.50	Low

Table 8 Health risk assessment for heavy metals in dust from EH, SP, LP and QP

			Cd	Cd_{canc}	Cr	Cr_{canc}	Cu	Mn	Ni	Ni_{canc}	Pb	Zn
		Ing RfD	1.00E-03		3.00E-03		4.00E-02	4.60E-02	2.00E-02		3.50E-03	3.00E-01
		Inh RfD	1.00E-03		2.86E-05		4.02E-02	1.43E-05	2.06E–02		3.52E-03	3.00E-01
		Dermal RfD	1.00E-03		6.00E-05		1.20E-02	1.84E-03	5.40E-03		5.25E-04	6.00E-02
		Inh SF		6.3E+00		4.20E+01				8.40E-01		
EH	Children	95% UCL	0.71	0.71	5.35	5.35	34.39	340.66	21.42	41.67	56.67	194.01
		HQ_{ing}	4.68E-03		1.17E-02		5.65E-03	4.87E–02	7.04E–03		1.06E–01	4.25E–03
		HQ_{inh}	1.31E-07		3.44E-05		1.57E-07	4.38E–03	1.91E–07		2.96E–06	1.19E–07
		HQ_{dermal}	2.62E-04		1.64E-03		5.28E-05	3.41E–03	7.30E–05		1.99E–03	5.95E–05
		$HI = \Sigma HQ_i$	4.94E-03		1.34E-02		5.71E-03	5.65E–02	7.12E–03		1.08E–01	4.31E–03
	Adults	HQ_{ing}	5.02E-04		1.26E-03		6.04E-04	5.22E–03	7.55E–04		1.14E–02	4.56E–04
		HQ_{inh}	7.38E-08		1.94E-05		8.86E-08	2.47E–03	1.08E–07		1.67E–06	6.70E–08
		HQ_{dermal}	2.00E-05		2.51E-03		8.06E-05	5.20E–03	1.12E–04		3.03E–03	9.09E–05
		$HI = \Sigma HQ_i$	5.22E-04		3.78E-03		6.86E-04	1.29E–02	8.66E–04		1.44E–02	5.47E–04
		LADD		3.65E–11		2.74E–10				1.10E–09		
		Cancer risk		2.30E–10		1.15E–08				9.23E–10		
SP	Children	95% UCL	1.43	1.43	15.53	15.53	62.87	306.13	31.06	31.06	98.06	394.45
		HQ_{ing}	9.41E–03		3.40E–02		1.03E–02	4.38E–02	1.02E–02		1.84E–01	8.65E–03
		HQ_{inh}	2.63E–07		9.97E–05		2.87E–07	3.93E–03	2.77E–07		5.12E–06	2.42E–07
		HQ_{dermal}	5.27E–04		4.76E–03		9.65E–05	3.06E–03	1.06E–04		3.44E–03	1.21E–04
		$HI = \Sigma HQ_i$	9.93E–03		3.89E–02		1.04E–02	5.08E–02	1.03E–02		1.88E–01	8.77E–03
	Adults	HQ_{ing}	1.01E–03		3.65E–03		1.11E–03	4.69E–03	1.09E–03		1.97E–02	9.26E–04
		HQ_{inh}	1.48E–07		5.62E–05		1.62E–07	2.22E–03	1.56E–07		2.89E–06	1.36E–07
		HQ_{dermal}	4.02E–05		7.27E–03		1.47E–04	4.68E–03	1.62E–04		5.25E–03	1.85E–04
		$HI = \Sigma HQ_i$	1.05E–03		1.10E–02		1.25E–03	1.16E–02	1.26E–03		2.50E–02	1.11E–03
		LADD		7.33E–11		7.96E–10				1.59E–09		
		Cancer risk		4.62E–10		3.34E–08				1.34E–09		
LP	Children	95% UCL	0.87	0.87	8.41	8.41	19.06	221.53	31.00	31.00	20.45	241.18
		HQ_{ing}	5.75E–03		1.84E–02		3.13E–03	3.17E–02	1.02E–02		3.84E–02	5.29E–03
		HQ_{inh}	1.61E–07		5.40E–05		8.71E–08	2.85E–03	2.77E–07		1.07E–06	1.48E–07
		HQ_{dermal}	3.22E–04		2.58E–03		2.92E–05	2.22E–03	1.06E–04		7.17E–04	7.40E–05
		$HI = \Sigma HQ_i$	6.08E–03		2.11E–02		3.16E–03	3.67E–02	1.03E–02		3.91E–02	5.36E–03
	Adults	HQ_{ing}	6.16E–04		1.97E–03		3.36E–04	3.39E–03	1.09E–03		4.12E–03	5.66E–04
		HQ_{inh}	9.06E–08		3.05E–05		4.91E–08	1.60E–03	1.56E–07		6.02E–07	8.33E–08
		HQ_{dermal}	2.46E–05		3.94E–03		4.47E–05	3.38E–03	1.61E–04		1.10E–03	1.13E–04
		$HI = \Sigma HQ_i$	6.41E–04		5.94E–03		3.80E–04	8.38E–03	1.25E–03		5.21E–03	6.79E–04
		LADD		4.49E–11		4.31E–10				1.59E–09		
		Cancer risk		2.83E–10		1.81E–08				1.34E–09		
QP	Children	95% UCL	1.29	1.29	13.98	13.98	72.77	302.28	23.87	23.87	284.29	453.40
		HQ_{ing}	8.46E–03		3.06E–02		1.20E–02	4.32E–02	7.85E–03		5.34E–01	9.94E–03
		HQ_{inh}	2.36E–07		8.98E–05		3.33E–07	3.88E–03	2.13E–07		1.48E–05	2.78E–07
		HQ_{dermal}	4.74E–04		4.29E–03		1.12E–04	3.02E–03	8.14E–05		9.97E–03	1.39E–04
		$HI = \Sigma HQ_i$	8.93E–03		3.50E–02		1.21E–02	5.01E–02	7.93E–03		5.44E–01	1.01E–02
	Adults	HQ_{ing}	9.06E–04		3.28E–03		1.28E–03	4.63E–03	8.41E–04		5.72E–02	1.06E–03
		HQ_{inh}	1.33E–07		5.06E–05		1.88E–07	2.19E–03	1.20E–07		8.37E–06	1.57E–07
		HQ_{dermal}	3.61E–05		6.55E–03		1.70E–04	4.62E–03	1.24E–04		1.52E–02	2.12E–04
		$HI = \Sigma HQ_i$	9.42E–04		9.88E–03		1.45E–03	1.14E–02	9.65E–04		7.25E–02	1.28E–03
		LADD		6.59E–11		7.16E–10				1.22E–09		
		Cancer risk		4.15E–10		3.01E–08				1.03E–09		

Table 9 Pearson's Correlation of heavy metals in dust at all recreational parks

	Cd	Cr	Cu	Mn	Ni	Pb	Zn
Cd	1.000						
Cr	0.592**	1.000					
Cu	0.134	0.684**	1.000				
Mn	0.122	0.013	0.207	1.000			
Ni	0.476*	0.725**	0.326	0.024	1.000		
Pb	0.181	0.575**	0.877**	0.300	0.293	1.000	
Zn	0.193	0.779**	0.832**	0.051	0.601**	0.793**	1.000

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

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

Table 10 Principal component analysis for heavy metals in dust at recreational parks

Variable	Rotated component matrix		
	PC1	PC2	PC3
Cd	–	0.890	0.200
Cr	0.625	0.720	–
Cu	0.947	0.112	0.115
Mn	0.125	–	0.965
Ni	0.331	0.784	− 0.135
Pb	0.906	–	0.248
Zn	0.903	0.316	− 0.105
Eigenvalues	3.832	1.329	1.021
Variability %	54.739	18.983	14.586
Cumulative %	54.739	73.722	88.308

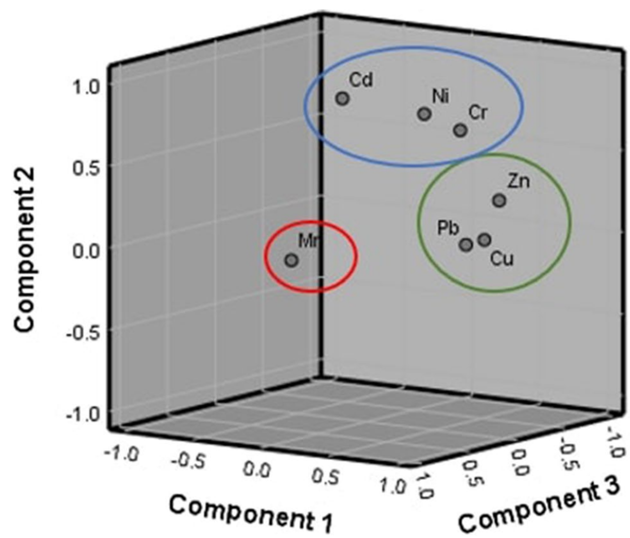

Fig. 4 Loading plot of PCA for heavy metals in dust at recreational parks

Figures 4 and 5, respectively, displays the loading plot of PCA and dendrogram of HCA for heavy metals in dust at all the recreational parks investigated. Both diagrams confirm the three groupings obtained for the data, with clusters 1 (Cu, Pb and Zn) and 2 (Cd, Cr and Ni) signifying anthropogenic (mainly vehicular) sources and cluster 3 representative of natural origin (Mn).

Conclusion

This comprehensive assessment revealed significant variation in heavy metal content attributed mainly to the changing land use across the locations. Based on indices that compare metal content with average background values (geoaccumulation index, pollution index, enrichment factor and ecological risk index) low to moderate degrees of pollution were recorded at the different parks, with higher degrees of contamination directly related to the increasing influence of traffic-related activities. Nevertheless, at the recreational parks investigated, the overall potential ecological risk due to heavy metal contamination was deemed to be low, and there were no significant non-carcinogenic and carcinogenic risks to children and adults based on exposure to dust at these locations. Multivariate statistical tools confirmed the apportionment of heavy metals in the dust samples, which were attributed mainly to vehicular-related origins, however, contributions from other sources of heavy metals cannot be ignored (natural occurrence, other anthropogenic activities). This study provided a robust evaluation of the heavy metal pollution status in dust at various recreational parks on the island of Trinidad. From the findings, the impact of traffic-related activities in the vicinity of these parks could potentially have significant impacts on the quality of the environment and the health of those who frequent these recreational sites. Thus, studies such as these demonstrate the importance of pollutant monitoring in urban environments. While effective and continuous monitoring may be necessary in designing policies for the appropriate management and preservative of these parks, an immediate solution may include

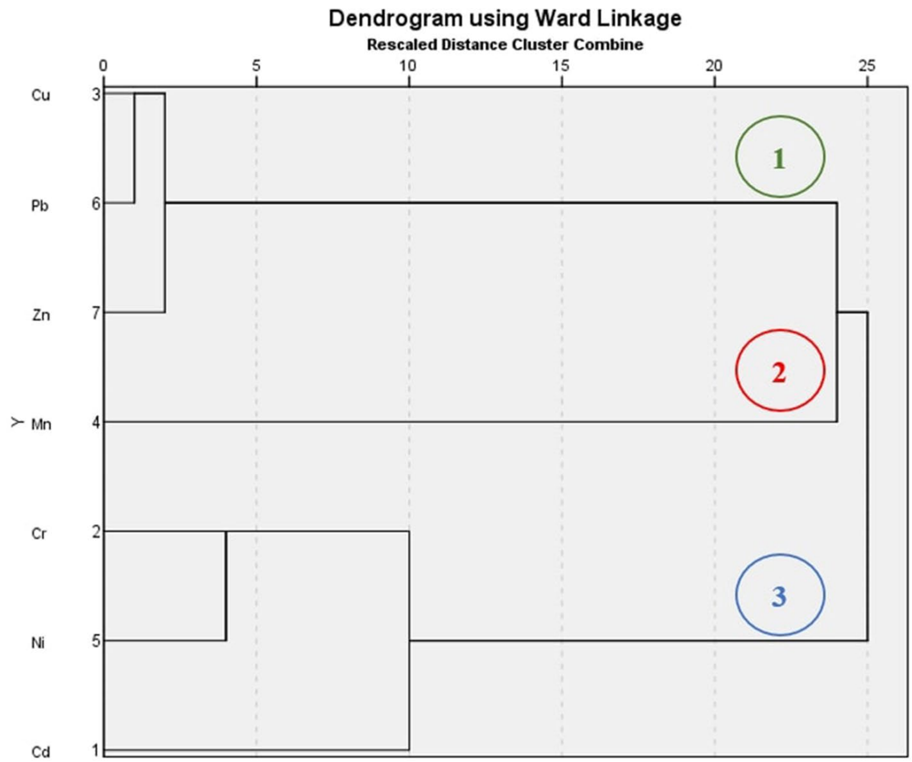

Fig. 5 Dendrogram of HCA for heavy metals in dust at recreational parks

implementation of physical barriers that mitigate traffic-related emissions on pedestrians and users of these parks.

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Declarations

Conflict of interest The authors declare no conflict of interest.

Ethics approval and consent to participate Exemption from Review was granted by the Campus Research Ethics Committee at the University of the West Indies, St. Augustine, Trinidad and Tobago, W.I.

Ethical responsibilities of authors All authors have read, understood, and have complied as applicable with the statement on "Ethical responsibilities of Authors" as found in the Instructions for Authors.

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