Environ Chem Lett (2014) 12:209–218
DOI 10.1007/s10311-013-0436-0

ORIGINAL PAPER

Human health risk from Pb in urban street dust in northern UK cities

Nwabueze I. Elom · Jane Entwistle · John R. Dean

Received: 21 August 2013 / Accepted: 4 September 2013 / Published online: 20 September 2013

Abstract The presence of Pb in the environment can cause significant health problems. These issues are exasperated when the lead is in a more amenable form for potential ingestion. This study investigates the potential human health risk from Pb in urban street dusts. The lead levels in urban street dust in major city centres in northern UK have been compared to levels determined in 35 cities around the world. With a few exceptions, it was noted that the mean Pb levels in this study exceeded those found in other cities worldwide. Samples ($n = 15$) of urban street dust were collected across five city centres, and specifically in areas in which pedestrians are likely to concentrate, as well as near historical buildings. Typical total lead concentrations across all sampling sites ranged from 306 to 558 mg/kg. The human health risk was assessed using oral bioaccessibility testing of the urban street dust. The mean oral bioaccessibility data, irrespective of site and sample location, were in the range 43 ± 9 %. The total and bioaccessible concentrations of lead were compared to the estimated tolerable daily intake (TDI_{oral}) values based on unintentional soil/dust consumption. It is noted, in all cases, that the maximum estimated lead daily intake exceeded the TDI_{oral}. An alternative approach for assessing the daily intake is proposed based on the actual measured air quality in selected cities on the same day as the sampled urban dust.

N. I. Elom · J. R. Dean (✉)
Department of Applied Sciences, Northumbria University, Newcastle upon Tyne NE1 8ST, UK
e-mail: John.Dean@northumbria.ac.uk

J. Entwistle
Department of Geography, Northumbria University, Newcastle upon Tyne NE1 8ST, UK

Keywords Lead · Oral bioaccessibility · Human risk assessment · Urban street dust · ICP-MS

Introduction

The term dust refers to minute solid particles emitted into the air from various sources and which are found to have settled onto outdoor objects and surfaces due to either wet or dry deposition (Ferreira-Baptisa and De Miguel 2005; Gill et al. 2006). Urban street dust in particular is a complex mixture consisting of suspended particles (atmospheric aerosol) and displaced soil and biogenic materials (e.g. tree leaves, debris and other plant matter) that can be easily mobilised by moving vehicles (Al-khashman 2007; Charlesworth et al. 2011; Fergusson and Kim 1991; Shi et al. 2008). In addition, emissions from a range of anthropogenic sources (e.g. vehicular exhausts particles, tyre wear, brake lining wear particles, municipal waste incineration as well as construction and building renovations) can all contribute to urban dust composition (Omar et al. 2007; Duzgoren-Aydin 2007; Han et al. 2008; Wei et al. 2010; Wei and Yang 2010; Manasreh 2010). Urban dusts have high-surface area and are easily transported and deposited thereby contributing significantly to potentially toxic elements (PTEs) load (Celis et al. 2004; Allout et al. 1990; Zhao et al. 2006; Irvine et al. 2009).

Dusts can be seen to pose more risk to human health when compared to other environmental matrices like soil; this is due to its pervasive and omnipresent nature (Banerjee 2003). Potential threats to human health due to elevated concentrations of PTEs in urban street dust are now well recognised (Schwar et al. 1988; Shi et al. 2010), and concerns have been expressed about the acute or prolonged long-term adverse effects on human health and

ecosystems in general (Lu et al. 2003; Lough et al. 2005; Steiner et al. 2007). In addition, the presence of PTEs in urban street dusts can have an adverse effect on air and water quality (Duong and Lee 2009).

Urban street dust could cause potentially adverse health effects since opportunities exist for a variety of different exposure pathways. For example, exposure could occur as a result of the association of dust particles with consumed food (e.g. eating outdoors) and dust particle adherence to hands followed by hand-to-mouth contact both leading to ingestion of urban street dust (Sezgin et al. 2004; Abrahams 2002) or alternatively inhalation. The inhalation pathway is not the subject of this paper though could be a significant contributor to human health risk, particularly on precipitation-free windy days when urban street dust will be at its most mobile (Laidlaw and Taylor 2011).

Children have the greatest risk considering their outdoor activities and their hand-to-mouth behaviour (Laidlaw and Filippelli 2008; Mielke et al. 1999; Shi et al. 2008). In addition, exposure via chronic contact of urban dust by children living within the vicinity of busy roads is also a significant pathway (Aelion et al. 2008). Numerous studies have been done on urban dusts specifically with respect to their PTE content, fractionation, source identification and contamination assessment, particle size and spatial distribution (e.g. Ahmed et al. 2007; Li et al. 2001; Duong and Lee 2011; Wei and Yang 2010; Han et al. 2008; Apeagyei et al. 2011; Wei et al. 2009; Charlesworth and Lees 1999; Mckenzie et al. 2008) as well as the contribution of lead-contaminated dust to children's blood lead levels (Liu et al. 2011). On the other hand, there is little information on the human health risk assessment via the ingestion pathway (i.e. oral bioaccessibility) of urban dust. Previous work from this group (Okorie et al. 2012) has determined the total and oral bioaccessible fraction of six elements in urban street dust from one study site. The research highlighted a particular concern for the high lead content of urban street dust. This paper extends the research further by considering the lead content of urban street dust in five cities, all relatively close to each other and each with a known historical past.

Experimental

Fifteen urban street dusts were collected (typically between 2 and 5 g per site) from each of the five different cities in the UK: Durham, Edinburgh, Liverpool, Newcastle upon Tyne and Sunderland. The samples were all collected in the summer of 2010. Newcastle upon Tyne samples were collected on the 27th and 28th May, Sunderland samples on the 31st May, Liverpool samples on the 5th June, Edinburgh samples on the 12th June and Durham samples on the 15th June. The sampling days were selected due to the lack of precipitation, i.e. they were dry and sunny. All the sampled sites were selected randomly but with due regard to the volume of traffic and the location of pedestrians, i.e. central city locations were selected which featured, if possible, pedestrian walkways. Dust samples were collected using a plastic dustpan and brush (Robertson and Taylor 2007; Zhang and Wang 2009). Different dustpans and brushes were used at each site; in addition, gloves were worn to avoid cross contamination. Collected samples were transferred to self-sealing Kraft bags for transportation back to the laboratory. The sampling procedure was maintained for all sites to minimise sampling variability and maintain sample integrity. The samples were dried in a drying cabinet at a temperature of 35 °C for 48 h. The dust samples were then sieved using a <125 μm nylon sieve to remove extraneous matter such as small pieces of building material and other debris. The <125 μm dust samples collected after sieving were weighed (their mass recorded) and stored in sealed plastic containers. All procedures of handling were carried out without contact with metal objects/utensils to avoid potential cross contamination of the samples. All chemicals used in analyses were certified analytical grade. Details of their sources have been reported elsewhere (Elom et al. 2013).

Procedure for sample extraction using UBM and microwave digestion

The in vitro extraction test employed in this work is based on the unified bioaccessibility method (UBM); the preparation of the reagents has been described elsewhere (Wragg et al. 2009) as well as the generic extraction protocols and microwave digestion procedure (Okorie et al. 2012).

Procedure for ICP-MS determination

Samples to be analysed by ICP-MS were prepared in triplicate by measuring 1 mL of either the filtrate, CRM/guidance material or blank into a 10-mL Sarstedt tube; this was followed by 30 μL of an internal standard (Terbium) and 9 mL of water (1 % HNO_3). The use of the CRM/guidance material was to assess the precision and accuracy of the methodology, whilst reagent blanks were included to check contamination. Calibration standards in the range 0–400 ppb (7 data points) were prepared, and internal standards were added; this was used to calibrate the instrument and also to construct the calibration graph. The instrument was tuned to verify mass resolution and maximise sensitivity; ^{208}Pb was used to determine the content of samples and standards. A calibration curve based on a concentration range of 0–400 ppb, with 7 calibration data points, was done, and the regression coefficient (R^2) was obtained (0.999).

Results and discussion

The total concentration of Pb in urban street dust, from the five cities, was determined (Table 1). The appropriate accuracy of the total determination was established by the analysis of CRM BCR 143R (Table 1). A box plot of the data (Fig. 1a) shows the variations been different sampling locations in all the studied cities. In Newcastle upon Tyne, the Pb concentration varied between 94 and 1,636 mg/kg with a mean concentration of 558 mg/kg. The highest Pb level in this city was obtained from site 3 (i.e. 1,636 mg/kg), i.e. a central focus point and pedestrianised area in the city centre (near Grey's Monument). In Durham, the concentration varied between 109 and 2,119 mg/kg with a mean concentration of 446 mg/kg; the highest concentration was determined on a popular street (Saddler Street) with both vehicles and pedestrians. In Liverpool, the concentration varied between 109 and 915 mg/kg with a mean concentration of 362 mg/kg; the highest concentration was found in a busy residential area (Brownlow Hill). In Edinburgh, the concentration varied from 76 to 1,273 mg/kg with a mean concentration of 443 mg/kg; the highest concentration was determined in the pedestrainized area close to the Palace of Holyrood house. Finally, in Sunderland, the concentration varied between 88 and 2,228 mg/kg with a mean concentration of 407 mg/kg; the highest was observed on a busy road (High Street West) close to the main shopping area.

A key characteristic of all the sites investigated, and especially those found to contain the highest concentrations, was their proximity to a high population density, i.e. their central location within or close to pedestrianised areas. Previous work from this group (Okorie et al. 2012) identified mean Pb levels in Newcastle upon Tyne of 992 mg/kg (range 70.2–4,261 mg/kg), with a mean Pb concentration across 26 cities worldwide of 463 mg/kg. In addition, previous data (Ross et al. 2007) from the UK Soil and Herbage Pollutant Survey identified mean Pb levels of 110 ± 90 mg/kg ($n = 87$) (with a range of 8.6–387 mg/kg) in soils from urban locations.

Although direct comparison of results from different studies is complicated due to the variation in sampling methods, the different sample preparation methodologies, the variety of particle size fractions adopted as well as the digestion and sample analysis protocols, the mean results obtained in this study compared favourably with results obtained from other studies from different cities worldwide (Table 2). The mean Pb concentrations obtained in this study were found to be higher than the mean concentration from 32 cities, whilst results from Lancaster (UK), London (UK) and Madrid (Spain) were ≥2× higher. The general similarity of results across all the cities suggests that the sources of Pb in the urban street dust could be traced to common urban sources.

Table 1 Summary of total, stage-related bioaccessibility and residual fraction of lead in urban street dusts and CRMs

CRM	Certified value (mg/kg)	Mean ± SD; ($n = 3$) (mg/kg)	In vitro gastrointestinal extraction mg/kg								
			Stage I (gastric digest)		Stage II (gastric + intestinal digest)			Stage III (residual digest)			
			Mean ± SD	% BAF	Mean ± SD	% BAF		Mean ± SD	% Recovery (stage II + III)		
BCR 143R	174 ± 5	173 ± 2	69.8 ± 2	40.3	42.7 ± 1	24.7		101 ± 3	83.2		
BGS 102	13 ± 6	NA	8.0 ± 4.0[a]	NA	NA	NA		NA	NA		
City (UK)	Mean (mg/kg) ($n = 15$) (minimum–maximum)	Minimum	Median	Maximum (% BAF)	Minimum	Median	Maximum (% BAF)	Minimum	Median	Maximum (% residual)	
Durham	446 ± 540 (109–2,119)	42.3	118	845 (39.9)	38.7	102	803 (37.9)	59.1	141	1,227 (57.9)	
Edinburgh	443 ± 346 (76–1,273)	29.5	136	617 (48.5)	21.1	108	584 (45.9)	39.6	154	643 (50.5)	
Liverpool	362 ± 197 (109–915)	34.2	143	374 (40.9)	28.7	121	293 (32.0)	43.2	153	412 (45.0)	
Newcastle upon Tyne	558 ± 556 (28–1,636)	9.10	121	736 (45.1)	8.1	109	534 (32.6)	14.5	143	934 (61.3)	
Sunderland	407 ± 593 (88–2,228)	45.4	83.1	895 (40.2)	35.8	69.3	687 (30.8)	42.3	91.4	1,211 (54.4)	

% BAF: stage-related bioaccessibility for the dust sample releasing the highest stage concentration, calculated as a fraction of that sample's total for both gastric and intestinal; % residual: residual fraction calculated as a fraction of the total for the sample releasing the highest residual concentration; NA = not available

[a] Mean value is based on repeat extracts generated over a period of 2 weeks and measured on 7 different occasions

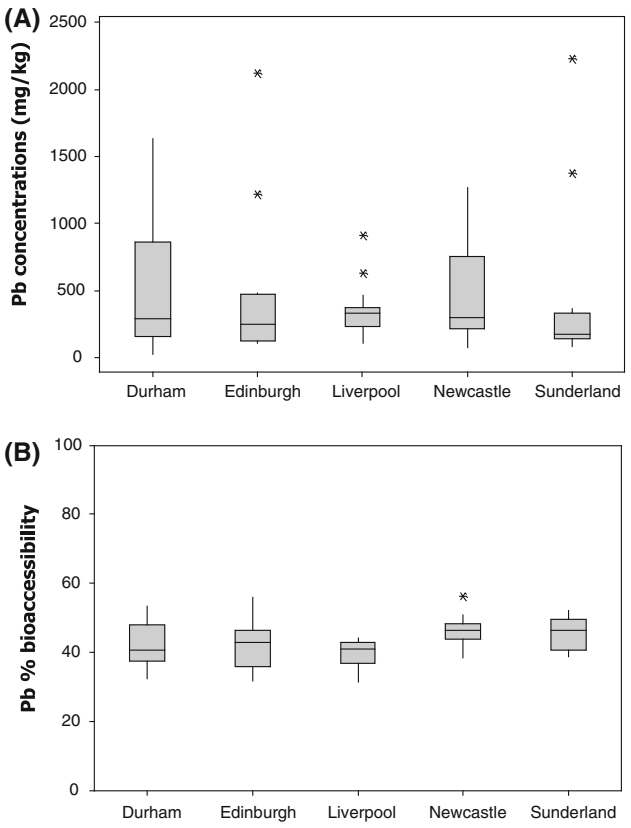

Fig. 1 Box plots of **a** total lead and **b** oral bioaccessibility (gastric only) of lead in urban street dusts showing median, mean, box boundary (25th and 75th) percentile and whiskers (10th and 90th) percentile

Oral bioaccessibility data can be used to estimate the amount of Pb, which could be absorbed into the body through the oral ingestion pathway; this is fundamental in assessing risks to humans particularly with respect to elevated Pb levels found in all the cities. The oral bioaccessibility of Pb from the 90 urban street dusts collected (from 5 cities) was assessed using the in vitro gastrointestinal extraction procedure (UBM). The results are shown in Table 1 as the minimum, median and maximum values.

However, the stage-related bioaccessibility (% BAF) is shown only for the maximum determined Pb concentration for each extraction stage thereby representing the worst-case scenario. The results show that the maximum % BAF was always highest in the gastric digest stage with an average of 43 ± 9 % (across all 5 cities). This is in line with many current findings where higher bioaccessibility values have been reported in the gastric phase (Lu et al. 2011; Poggio et al. 2009; Turner 2011). A box plot, showing the mean, median, box boundary (25th and 75th) percentile and whiskers (10th and 90th) percentile, summarises each individual %BAF result (Fig. 1b). In addition, the two certified reference materials (BGS Guidance Material 102 and BCR 143R) were also subjected to the same procedure. The results (Table 1) show reasonable agreement for the certified oral bioaccessibility guidance material (BGS 102); data are also reported for BCR143R showing a total recovery of 83.2 % based on stages II and II. This indicates that the analytical procedures in place were able to be replicated appropriately.

In order to estimate the human health risk associated with exposure to urban street dust, the concentration of Pb from a particular sample that a child (as the most sensitive receptor) might possibly ingest to reach the estimated tolerable daily intake (TDI, for oral ingestion) can be calculated (Pouschat and Zagury 2006; Swartjes 2011; Okorie et al. 2012) (Table 3).

Comparing the data in Table 3 with the TDI_{oral} (Baars et al. 2001) for Pb (3.6 $\mu g/kg_{bw}$/day), it is observed that all locations exceed the calculated maximum daily intake, irrespective of city. Typically, a child (aged between 1 and <6 years) would need to consume 32 mg (Durham), 53 mg (Edinburgh), 73 mg (Liverpool), 41 mg (Newcastle) and 30 mg (Sunderland) per day in order to exceed the TDI_{oral} guidelines. Minimum daily intake values are also included in Table 3; in each of these cases, the TDI_{oral} is never exceeded. However, in any risk assessment, it is appropriate to always consider the worst-case scenario, i.e. based on maximum TDI_{oral}. In addition, it is also possible to consider the actual exposure frequency that a child may be exposed to urban dust as a result of anticipated visits to the city centres over the period of 12 months (Table 3). By including exposure frequency, it is noted that the maximum estimated daily intake is considerably reduced (i.e. <0.50 $\mu g/kg_{bw}$/day) and over 7 times lower than the TDI_{oral}.

Also, based on the determined oral bioaccessibility (gastric phase only), it is possible to modify the equation to additionally include the fractional bioavailability (Okorie et al. 2012) as a numerator. The results, Table 3, for the maximum daily intake are marginally in excess of the TDI_{oral} in three locations only (Durham, Newcastle and Sunderland); note that, minimum daily intakes are also included for comparison. The gastric phase was used to compute bioaccessible TDI's because it yielded higher bioaccessibility in all cases and so represents the higher risk. However, it is not expected that a child would ingest (even one with pica behaviour) 100 mg of urban street dust during a 1 h/day visit (throughout the year) to these city locations.

Whilst the data have been based on a daily ingestion rate of 100 mg/day (U.S. EPA 2008), it is believed that this is an overestimate in the context of urban street dust ingestion. We therefore suggest (Okorie et al. 2012) that a realistic assessment of human health risks should be based on the actual determined airborne particulate matter. The process proposed is also limited by some assumptions:

Table 2 Examples of Pb concentrations in street dust from across the world

City, Country (year of study) (number of samples)	Pb[a] (mg/kg)	Comments	References
Europe			
Lancaster, UK (1979)	1,740	Mean values from six different areas (car parks, garage forecourts, town centre streets, main roads, residential areas and rural roads). <30-μm particle size was used.	Harrison (1979)
London, UK (1988)	1,030	Mean values from urban locations	Schwar et al. (1988)
Coventry, UK (2003) ($n = 49$)	47.1 ± 8.4 (0.0–199)	Mean values from 3 locations (industrial, residential and parks), particle size, <1 mm	Charlesworth et al. (2003)
Birmingham, UK (2003) ($n = 49$)	48.0 ± 2.9 (0.0–146)	Mean values from 3 locations (industrial, residential and parks), particle size, <1 mm	Charlesworth et al. (2003)
Gela, Italy (2006) ($n = 8$)	60	Mean values from 3 locations (industrial, urban and peripheral) and 6 particle size ranges (500–250, 250–125, 125–63, 63–40, 40–20, <20 μm). No standard deviation and range reported. Mean values from the urban location and in the particle size of 125–63 μm were reported on the table above as this tally with the current work.	Manno et al. (2006)
Kavala, Greece (2009) ($n = 96$)	387 ± 245 (75–2,500	Mean values from 3 areas (urban, industrial and peripheral), particle size (<63 μm). Mean value from the urban site shown.	Christoforidis and Stamatis (2009)
Oslo, Norway (1997) ($n = 16$)	180 ± 14	Mean values from urban surfaces (sidewalk, road and gutter), particle size <100 μm.	De Miguel et al. (1997)
Madrid, Spain (1997) ($n = 16$)	1,927 ± 508	Mean values from urban surfaces (sidewalk, road and gutter), particle size <100 μm.	De Miguel et al. (1997)
Yozgat, Turkey (2003) ($n = 45$)	56.0 ± 7.0 (47.0–62.6)	Mean vales from moderate traffic zones. Particle size fraction <30 μm.	Divrikli et al. (2003)
Kayseri, Turkey (2006) ($n = 29$)	74.8 ± 53.8 (27.9–312)	Mean values from urban streets, particle size <74 μm.	Tokalioglu and Kartal (2006)
North America			
Ottawa, Canada (2001) ($n = 45$)	39.1 (12.6–122)	Mean values from urban residential areas, particle size 100–124, 75–100 μm.	Rasmussen et al. (2001)
Massachusetts, USA (2011) ($n = 85$)	73 ± 181 (0–1,639)	Mean values from 3 locations (Fenway, Somerville and Greenfield); no particle size reported.	Apeagyei et al. (2011)
Florida, USA (2009) ($n = 200$)	18.3 ± 32.5 (<1.43–386)	Mean values from 20 municipalities, particle size 4.75 mm.	Jang et al. (2009)
Hermosillo, Mexico (2007) ($n = 25$)	36.15	Mean values from 3 locations (industrial, heavy and low traffic density, and commercial). Particle size 200 μm. Street dust property (LOI %) reported.	Meza-Figueroa et al. (2007)
Africa			
Gwagwalada, Nigeria (2005) ($n = 75$)	210 ± 16.4	Mean values from an urban street. Particle size 2 mm.	Mashi et al. (2005)
Mubi, Adamawa, Nigeria (2007) ($n = 10$)	20.4 ± 2.00	Mean values from 5 locations (mechanical workshops, motor parks, market areas, roundabouts/highways and residential areas). Particle size <250 μm.	Shinggu et al. (2007)
Luanda, Angola (2005) ($n = 92$)	351 ± 237 (74–1,856)	Mean values from city centre and urban residential areas; particle size <100 μm was used.	Ferreira-Baptisa and De Miguel (2005)
Asia			

Table 2 continued

City, Country (year of study) (number of samples)	Pb[a] (mg/kg)	Comments	References
Singapore (2009)	68.6 ± 25.9	Mean values from 3 locations (residential, commercial and industrial). Particle size used 355 μm.	Joshi et al. (2009)
Ulsan, Korea (2009) ($n = 12$)	136 ± 37	Mean values 3 locations (non-ferrous metal, petrochemical and mechanical/shipbuilding) particle sizes <75, 75–180, 180–850 and 850–2,000 μm.	Duong and Lee (2009)
Ulsan, Korea (2011) ($n = 12$)	92.1 ± 12.3	Mean values from 3 locations (traffic rotary stations, circulation road, highway and downtown). Particle size fractions <75, 75–180, 180–850 and 850–2,000 μm.	Duong and Lee (2011)
Taejon, Korea (1998) ($n = 81$)	52 (13–161)	Mean values from 2 locations (traffic and industrial areas) and <180-μm particle size fraction.	Kim et al. (1998)
Dhahran, Saudi Arabia (2010) ($n = 5$)	40.3 ± 34.9	Mean values from a residential camp and particle size <63 μm.	Turner and Hefzi (2010)
Calcutta, India (1999) ($n = 12$)	536 ± 39	Mean values from a residential site and <30-μm particle size fraction was used.	Chatterjee and Banerjee (1999)
Dhanbad and Bokaro, India (2011) ($n = 13$)	48 ± 29 (17–128)	Mean values from 4 zones (rural, urban, mining and industrial): particle size fractions used were 700, 500, 250, 125 and 63 μm.	Singh (2011)
Dhaka, Bangladesh (2006) ($n = 33$)	35 ± 7.78	Mean values from 3 zones (commercial, residential and commercial).	Ahmed and Ishiga (2006)
Islamabad, Pakistan (2009) ($n = 13$)	104 ± 29 (60–150)	Mean values from a busy expressway and <120-μm particle size fraction used.	Faiz et al. (2009)
Aqaba, Jordan (2007) ($n = 140$)	138 ± 6.2	Mean values from one zone (heavy, moderate and low traffic). <2-mm particle size fraction.	Al-khashman (2007)
Mutah, Jordan (2010) ($n = 24$)	143 ± 109 (26–368)	Mean values from one zone (major roads) and four particle fractions (2 mm, 500, 250 and 64 μm).	Manasreh (2010)
Xi'an, China (2006) ($n = 65$)	231 ± 431 (29–3,060)	Mean values from one zone (major roads and streets) and 66-μm particle size.	Yongming et al. (2006)
Urumqi, China (2010) ($n = 42$)	61.1 ± 14.1	Mean values from 4 zones (ring road, city centre and freeway).	Wei et al. (2010)
Beijing, China (2008) ($n = 25$)	61 ± 17	Mean values three zones (central, city centre and forbidden) and two particle size fractions <63 and 100 μm.	Tanner et al. (2008)
Shanghai, China (2008) ($n = 25$)	148 ± 58	Mean values three zones (central, city centre and forbidden) and two particle size fractions <63 and 100 μm.	Tanner et al. (2008)
Hong Kong, China (2008) ($n = 25$)	240 ± 73	Mean values three zones (central, city centre and forbidden) and two particle size fractions <63 and 100 μm.	Tanner et al. (2008)
Baoji, China (2004)	408 ± 295 (141–1,847)	Mean values from four areas (industrial, commercial, heavy and low density) and <75-μm particle size fraction.	Celis et al. (2004)
Nanjing, China (2011) ($n = 35$)	103 ± 48 (37.3–204)	Mean values from eight urban districts with different land uses one particle size fraction <63 μm.	Hu et al. (2011)

[a] Mean concentration (mg/kg) with the range values (minimum–maximum)

Table 3 Estimated lead daily intake (orally ingested) from urban dust

City (UK)	Estimated daily intake (μg/kg_{bw}/day)[a] based on 100 mg/day ingestion[d]		Estimated daily intake (ng/kg_{bw}/day)[b] based on 100 mg/day ingestion and an estimated annual exposure frequency[d]		Estimated daily intake (μg/kg_{bw}/day)[c] based on 100 mg/day ingestion and bioaccessibility[d,e]		Estimated daily intake (ng/kg_{bw}/day)[a] based on 100 mg/day ingestion and actual air quality data[d,f]	
	Minimum	Maximum	Minimum	Maximum	Minimum	Maximum	Minimum	Maximum
Durham	0.59	11.4	24.6	479	0.23	4.5	NA	NA
Edinburgh	0.41	6.8	17.2	288	0.20	3.3	0.04[g]	0.62[g]
Liverpool	0.59	4.9	24.6	207	0.24	2.0	0.22[h]	1.87[h]
Newcastle	0.15	8.8	6.3	369	0.07	4.0	0.01[i]	0.70[i]
Sunderland	0.47	12.0	19.9	503	0.19	4.8	NA	NA

NA not available

[a] The maximum estimated daily intake (DI) was calculated using the formula DI = [EC × SIR]/BW, where DI = daily intake (μg/kg_{bw}/day) as determined in <125 μm fraction of the soil sample with the highest concentration; EC = exposure concentration of Pb in <125 μm (μg/g); SIR = soil ingestion rate (0.05 g/day) (U.S. EPA 2008); and BW = body weight (18.6 kg for a 3–6-year old child) (adapted from Pouschat and Zagury 2006)

[b] The maximum estimated daily intake (DI) was calculated using the formula DI = [EC × SIR × EF]/BW, where DI = daily intake (ng/kg_{bw}/day) as determined in <125 μm fraction of the soil sample with the highest concentration; EC = Exposure concentration of Pb in <125 μm (μg/g); SIR = soil ingestion rate (0.05 g/day) (U.S. EPA 2008); EF = exposure frequency = 0.042 (estimated to be 7 h a week for 52 weeks, i.e. 364 h/year or 15.2 days/year); and BW = body weight (18.6 kg for a 3–6-year old child) (adapted from Pouschat and Zagury 2006)

[c] The maximum estimated daily intake (DI) was calculated using the formula DI = [EC × SIR × B]/BW, where DI = daily intake (μg/kg_{bw}/day) as determined in <125 μm fraction of the soil sample with the highest concentration; EC = exposure concentration of Pb in <125 μm (μg/g); SIR = soil ingestion rate (0.05 g/day) (U.S. EPA 2008); B = oral bioaccessibility (where 100 % oral bioaccessibility is 1.0); and BW = body weight (18.6 kg for a 3–6-year old child) (adapted from Pouschat and Zagury 2006)

[d] Based on an assumption of a daily (soil + dust) ingestion rate of 100 mg (guideline value based on unintentional consumption) (U.S. EPA 2008)

[e] Based on the maximum bioaccessible concentration using either 'gastric' or 'gastric' + 'intestinal' phase

[f] Based on a determined air quality measurement in specific city centres between 10.00 and 17.00 h

[g] Air quality measurement in Edinburgh Centre on 12 June 2010 of 9 μg/m^3 air particulate matter (PM10) (http://uk-air.defra.gov.uk/data), and ingestion rate of 9 μg has been assumed as the child is likely to occupy an active volume of 1 m^3

[h] Air quality measurement in Liverpool Centre on 5 June 2010 of 38 μg/m^3 air particulate matter (PM10) (http://uk-air.defra.gov.uk/data), and ingestion rate of 38 μg has been assumed as the child is likely to occupy an active volume of 1 m^3

[i] Air quality measurement in Newcastle City Centre between 27 and 28 May 2010 of 8 μg/m^3 air particulate matter (PM10) (http://uk-air.defra.gov.uk/data), and ingestion rate of 8 μg has been assumed as the child is likely to occupy an active volume of 1 m^3

most notably that the data are specific to the actual date of sample collection per site; exposure duration of 5 h, i.e. between 10.00 and 15.00 h; that a child would occupy an active area of 1 m^3 (based on an estimate of hand-to-mouth distances); and that the urban dust particle size fraction is based on airborne dust (10 μm) [the samples analysed in this study were actually settled street dust (<125 μm)]. [Note: these particular particle size fractions (<10 and 125 μm) have the potential to easily enter the human body either through oral (mouth) or inhalation (nose/mouth) pathways]. It has been reported (Plumlee et al. 2010) that the relationship between the two environmental matrices is basically in terms of sources, physical and chemical characteristics as well as toxicological mechanisms in exposed populations. A recent study (Sipos et al. 2012) that investigated the concentration of Pb and Zn in airborne dust (PM_{10}) and settled dust described both environmental matrices as 'mobile toxic components' with the potential to harbour PTEs. The study highlighted that studies on sources, composition, distribution and health impacts of airborne dust and settled dust are necessary for their risk assessment to atmospheric quality, ecology and human health particularly in a populated urban environment.

In the UK, daily air quality data are published (http://uk-air.defra.gov.uk/data) providing experimental values for selected cities including Edinburgh, Liverpool and Newcastle upon Tyne; no data exist for Durham and Sunderland. The determined average experimental data (10-μm particle size, PM_{10}) available on the sampling dates were 8 μg/m^3 (Newcastle upon Tyne), 38 μg/m^3 (Liverpool) and 9 μg/m^3 (Edinburgh). It is then possible to reapply the equation using a revised dust ingestion rate of the following: 0.008 mg/day (Newcastle upon Tyne); 0.038 mg/day (Liverpool) and 0.009 mg/day (Edinburgh). It is now noted that the maximum Pb daily intake is considerably reduced (Table 3) compared to the TDI_{oral}; it is concluded that the risk from Pb in urban dust is minimal when including the actual air quality data. However, it is to be noted that

TDI_{oral} only calculated from street dust can be used to assess health, but it was considered necessary to also carry out air quality calculations as a conservative and a holistic approach to human health risk assessment.

Conclusions

By considering the worst-case scenario based on maximum determined Pb concentrations in urban dust across five cities has indicated the low risk associated with exposure. Whilst the maximum estimated daily intake from Pb in urban dust exceeds the TDI_{oral} in all cases, it is an over-estimation of risk to the child. By including the oral bio-accessibility into the data analysis, the results indicate that the amount of Pb which is available for absorption into the body, through the oral ingestion pathway, is approximately 60 % lower. The use of oral bioaccessibility data is therefore fundamental in assessing risks to humans particularly with respect to elevated Pb levels found in this study and previously (Okorie et al. 2012).

Given the complexity of modelling exposure and intake pathway (ingestion/inhalation/dermal sorption), particularly in an urban environment, our risk assessments based on worst-case scenarios (i.e. maximum concentrations) must be treated with caution. Furthermore, whilst it is not expected that a child would ingest 100 mg (accepted soil + dust ingestion rate, U.S. EPA 2008) of urban dust during a daily visit to the city centres of these cities, our study shows that a child only needs to ingest <73 mg dust/day in order to exceed the Pb TDI_{oral}. This coupled with the 43 % bioaccessibility highlights the need for regular monitoring of Pb levels in the urban environment and of robust environmental management practices, including regular street sweeping. With the gradual move from leaded to unleaded petrol, a number of studies have observed a reduction in Pb concentration in urban street dusts over recent decades (e.g. Charlesworth et al. 2003; De Miguel et al. 1997; Duzgoren-Aydin 2007; Laidlaw and Filippelli 2008); however, despite this decline monitoring of Pb as part of atmospheric monitoring programs remains warranted along with environmental policies (i.e. air quality management zones) to target a reduction in human exposure to Pb in the urban environment.

Alternate approaches for assessing the maximum daily Pb intake from urban dust have also been considered including those based on the likely annual exposure frequency and the actual air quality data (from three cities). In all cases, using these revised approaches, the environmental health risk was considerably reduced.

Acknowledgments We are grateful to Ebonyi State University Abakaliki/Education Trust Fund, Nigeria, for funding the research. The technical assistance of Mr. Gary Askwith in the operation of the ICP-MS is highly appreciated.

References

Abrahams PW (2002) Soils: their implications to human health. Sci Total Environ 291:1–32. doi:10.1016/S0048-9697(01)01102-0

Aelion CM, Davies HT, McDermott S, Lawson AB (2008) Metal contamination in rural topsoil in South Carolina: potential for human health impact. Sci Total Environ 402:149–156. doi:10.1016/j.scitotenv.2008.04.043

Ahmed F, Ishiga H (2006) Trace metal concentrations in street dusts of Dhaka city, Bangladesh. Atmos Environ 40:3835–3844. doi:10.1016/j.atmosenv.2006.03.004

Ahmed F, Hawa Bibi M, Ishiga H (2007) Environmental assessment of Dhaka City (Bangladesh) based on trace metal contents in road dust. Environ Geol 51:975–985. doi:10.1007/s00254-006-0367-1

Al-Khashman OA (2007) The investigation of metal concentrations of street dust samples in Aqaba city, Jordan. Environ Geochem Health 29:197–207. doi:10.1007/s10653-006-9065-x

Allout RW, Hewitt CN, Kelly MR (1990) The environmental half-lives and mean residence times of contaminants in dust for an urban environment: Barrow-in-Furness. Sci Total Environ 93:403–410. doi:10.1016/0048-9697(90)90131-D

Apeagyei E, Bank MS, Spengler JD (2011) Distribution of heavy metals in road dust along urban-rural gradient Massachusetts. Atmos Environ 45:2310–2323. doi:10.1016/j.atmosenv.2010.11.015

Baars AJ, Theelen RMC, Janssen PJCM, Hesse JM, van Apeldoorn ME, Meijerink MCM, Verdam L, Zeilmaker MJ (2001) Re-evaluation of human toxicological maximum permissible risk levels. National Institute of Public Health and the Environment (RIVM), Report 711701025

Banerjee ADK (2003) Heavy metal levels and solid phase speciation in street dusts of Delhi, India. Environ Pollut 123:95–105. doi:10.1016/S0269-7491(02)00337-8

Celis JE, Morales JR, Zaror CA, Inzunza JC (2004) A study of the particulate matter PM10 composition in the atmosphere of Chillan, Chile. Chemosphere 54:541–550. doi:10.1016/S0045-6535(03)00711-2

Charlesworth SM, Lees JA (1999) Particulate-associated heavy metals in urban environment: their transport from source to deposit, Coventry, UK. Chemosphere 39:833–848. doi:10.1016/S0045-6535(99)00017-X

Charlesworth SM, Everett M, McCarthy R, Ordonez A, De Miguel E (2003) A comparative study of heavy metal concentration and distribution in deposited street dusts in a large and small urban area: Birmingham and Coventry, West Midlands, UK. Environ Int 29:563–573. doi:10.1016/S0160-4120(03)00015-1

Charlesworth S, DeMiguel E, Ordonez A (2011) A review of the distribution of particulate trace elements in urban terrestrial environments and its application to consideration of risk. Environ Geochem Health 33:103–123. doi:10.1007/s10653-010-9325-7

Chatterjee A, Banerjee RN (1999) Determination of lead and other metals in a residential area of greater Calcutta. Sci Total Environ 227:175–185. doi:10.1016/S0048-9697(99)00026-1

Christoforidis A, Stamatis N (2009) Heavy metal contamination in street dust and road side along the major national road Kavala's region, Greece. Geoderma 151:257–263. doi:10.1016/j.geoderma.2009.04.016

De Miguel E, Llamas JF, Chacon E, Berg T, Larssen S, Royset O, Vadset M (1997) Origin and patterns of distribution of trace

elements in street dust: unleaded petrol urban lead. Atmos Environ 31:2733–2740. doi:10.1016/S1352-2310(97)00101-5

Divrikli U, Soylak M, Elci L, Dogan M (2003) Trace heavy metal levels in street dusts samples from Yozgat city center, Turkey. J Trace Microprobe Tech 21:351–361. doi:10.1081/TMA-120020270

Duong TTT, Lee B-K (2009) Partitioning and mobility behaviour of metals in road dust from national-scale industrial areas in Korea. Atmos Environ 43:3502–3509. doi:10.1016/j.atmosenv.2009.04.036

Duong TTT, Lee B-K (2011) Determining contamination level of heavy metals in road dusts from busy traffic areas with different characteristics. J Environ Manag 92:554–562. doi:10.1016/j.jenvman.2010.09.010

Duzgoren-Aydin NS (2007) Sources and characteristics of lead pollution in the urban environment of Guanzhou. Sci Total Environ 385:182–195. doi:10.1016/j.scitotenv.2007.06.047

Elom NI, Entwistle JA, Dean JR (2013) How safe is the playground? An environmental health risk assessment of As and Pb levels in playing fields in NE England. Environ Chem Lett. doi:10.1007/s10311-013-0413-7

Faiz Y, Tufail M, Tayyeb Javed M, Chaudhry MM, Siddique N, Lee Y-L (2009) Road dust pollution of Cd, Cu, Ni and Zn along Islamabad Expressway, Pakistan. Microchem J 92:186–192. doi:10.1016/j.microc.2009.03.009

Fergusson JE, Kim ND (1991) Trace elements in house and street dusts: sources and speciation. Sci Total Environ 100:125–150. doi:10.1016/0048-9697(91)90376-P

Ferreira-Baptisa L, De Miguel E (2005) Geochemistry and risk assessment of street dust in Luanda, Angola. Atmos Environ 39:4501–4512. doi:10.1016/j.atmosenv.2005.03.026

Gill ET, Zobeck TM, Stout JE (2006) Technologies for laboratory generation of dust from geological materials. J Hazard Mater 132:1–13. doi:10.1016/j.jhazmat.2005.11.083

Han Y, Cao J, Posmentier ES, Fung K, Tian H, An Z (2008) Particulate-associated potentially harmful elements in urban road dusts in Xi'an China. Appl Geochem 23:835–845. doi:10.1016/j.apgeochem.2007.09.008

Harrison RM (1979) Toxic metals in street and household dusts. Sci Total Environ 11:89–97. doi:10.1016/0048-9697(79)90036-6

Hu X, Zhang Y, Luo J, Wang T, Lian H, Ding Z (2011) Bioaccessibility and health risk of arsenic, mercury, and other metals in urban street dusts from a mega-city, Nanjing, China. Environ Pollut 159:1215–1221. doi:10.1016/j.envpol.2011.01.037

Irvine KN, Perrelli MF, Ngoen-Klan R, Droppo IG (2009) Metal levels in street sediment from an industrial city: spatial trends, chemical, and management implications. J Sci Sediment 9:328–341. doi:10.1007/s11368-009-0098-5

Jang Y, Jain P, Tolaymat T, Dubey B, Townshend T (2009) Characterization of pollutants in Florida street sweepings for management and reuse. J Environ Manag 91:320–327. doi:10.1016/j.jenvman.2009.08.018

Joshi UM, Vijayaraghavan K, Balasubramanian R (2009) Elemental composition of urban street dusts and their dissolution characteristics in various aqueous media. Chemosphere 77:526–533. doi:10.1016/j.chemosphere.2009.07.043

Kim KW, Myung JH, Ahn JS, Chon H-T (1998) Heavy metal contamination in dusts and stream sediments in the Taejon area, Korea. J Geochem Explor 64:409–419. doi:10.1016/S0375-6742(98)00045-4

Laidlaw MAS, Filippelli GM (2008) Resuspension of urban soils as a persistent source of lead poisoning in children: a review and new directions. Appl Geochem 23:2021–2039. doi:10.1016/j.apgeochem.2008.05.009

Laidlaw MAS, Taylor MP (2011) Potential for childhood lead poisoning in the inner cities of Australia due to lead in soil dust. Environ Pollut 150:1–9. doi:10.1016/j.envpol.2010.08.020

Li X, Poon CS, Liu PS (2001) Heavy metal contamination of urban soils and street dusts in Hong Kong. Appl Geochem 16:1361–1368. doi:10.1016/S0883-2927(01)00045-2

Liu J, Ai Y, McCauley L, Pinto-Martin J, Yan C, Shen X, Neddleman H (2011) Blood lead levels and associated sociodemographic factors among preschool children in the South Eastern region of China. Paediatr Perinatal Epidemiol 26:61–69. doi:10.1111/j.1365-3016.2011.01234.x

Lough GC, Schauer JJ, Park J-S, Shafer MM, DeMinter JT, Weinstein JP (2005) Emission of metals associated with motor vehicle roadways. Environ Sci Technol 39:826–836. doi:10.1021/es048715f

Lu Y, Gong Z, Zhang G, Burghardt W (2003) Concentrations and chemical speciation of Cu, Zn, Pb and Cr of urban soils in Nanjing, China. Geoderma 115:101–111. doi:10.1016/S0016-7061(03)00079-X

Lu Y, Yin W, Huang L, Zhang G, Zhao Y (2011) Assessment of bioaccessibility and exposure risk of arsenic and lead in urban soils of Guangzhou City, China. Environ Geochem Health 33:93–102. doi:10.1007/s10653-010-9324-8

Manasreh WA (2010) Assessment of trace metals in street dust of Mutah city, Karak, Jordan. Carpathian J Earth Environ Sci 5:5–12

Manno E, Varrica D, Dongarra G (2006) Metal distribution in road dust samples collected in an urban area close to a petrochemical plant at Gela, Sicily. Atmos Environ 40:5929–5941. doi:10.1016/j.atmosenv.2006.05.020

Mashi SA, Yaro SA, Eyong PN (2005) A survey of trends related to the contamination of street dust by heavy metals in Gwagwalada, Nigeria. Manag Environ J 16:71–76. doi:10.1108/14777830510574353

Mckenzie ER, Wong CM, Green PG, Kayhanian M, Young TM (2008) Size dependent elemental composition of road-associated particles. Sci Total Environ 398:145–153. doi:10.1016/j.scitotenv.2008.02.052

Meza-Figueroa D, De La O-Villanueva M, De la Parra ML (2007) Heavy metal distribution in dust from elementary schools in Hermosillo, Sonora, Mexico. Atmos Environ 41:276–288. doi:10.1016/j.atmosenv.2006.08.034

Mielke HW, Gonzales CR, Smith MK, Mielke PW (1999) The urban environment and the children's health: soil as an integrator of lead, zinc and cadmium in New Orleans, Louisiana, USA. Environ Res 81:117–129. doi:10.1006/enrs.1999.3966

Okorie A, Entwistle J, Dean JR (2012) Estimation of daily intake of potentially toxic elements from urban street dust and the role of oral bioaccessibility testing. Chemosphere 86:460–467. doi:10.1016/j.chemosphere.2011.09.047

Omar NYMJ, Abas MRB, Rahman NA, Tahir NM, Rushdi AI, Simoneit BRT (2007) Levels and distributions of organic source tracers in air and road side dust particles of Kuala Lumpur, Malaysia. Environ Geol 52:1485–1500. doi:10.1007/s00254-006-0593-6

Plumlee G, Wolf RE, Morman SA, Meeker GP, Durant JT, Neri A, Dooyema CA (2010) Mineralogical and geochemical influences on the 2010 Nigeria lead poisoning outbreak linked to artisanal gold ore processing. Geol Soc Am Abstr Program 42:354

Poggio L, Borut V, Schulin R, Hepperle E, Marsan FA (2009) Metals pollution and human bioaccessibility of topsoil in Grugliasco (Italy). Environ Pollut 157:680–689. doi:10.1016/j.envpol.2008.08.009

Pouschat P, Zagury GJ (2006) In vitro gastrointestinal bioavailability of arsenic in soils collected near CCA-treated utility poles. Environ Sci Technol 40:4317–4323. doi:10.1021/es0604156

Rasmussen PE, Subramanian SK, Jessiman BJ (2001) A multi-element profile of house dust in relation to exterior dust and soils in the city of Ottawa, Canada. Sci Total Environ 267:125–140. doi:10.1016/S0048-9697(00)00775-0

Robertson DJ, Taylor KG (2007) Temporal variability of metal contamination in urban road-deposited sediment in Manchester UK: implications for urban pollution monitoring. Water Air Soil Pollut 186:209–220. doi:10.1007/s11270-007-9478-x

Ross SM, Wood MD, Copplestone D, Warriner M, Crook P (2007) UK Soil and Herbage Pollutant Survey Report no. 7, Environment Agency

Schwar MJR, Moorcroft JS, Laxen DPH, Thompson M, Amorgie C (1988) Baseline metal-in-dust concentrations in Greater London. Sci Total Environ 68:25–43. doi:10.1016/0048-9697(88)90359-2

Sezgin N, Ozcan H, Demir C, Nemlioglu S, Bayat C (2004) Determination of heavy concentrations in street dusts in Istanbul E-5 highway. Environ Int 29:979–985. doi:10.1016/S0160-4120(03)00075-8

Shi G, Chen Z, Xu S, Zhang J, Wang L, Bi C, Teng J (2008) Potentially toxic metal contamination of urban soils and roadside dusts in Shanghai, China. Environ Pollut 156:251–260. doi:10.1016/j.envpol.2008.02.027

Shi G, Chen Z, Bi C, Li J, Teng J, Xu X (2010) Comprehensive assessment of toxic metals in urban and suburban street deposited sediments (SDSs) in the biggest metropolitan area of China. Environ Pollut 158:694–703. doi:10.1016/j.envpol.2009.10.020

Shinggu DY, Ogugbuaja VO, Barminas JT, Toma I (2007) Analysis of street dust for heavy metal pollutants in Mubi, Adamawa State, Nigeria. Int J Phys Sci 2:290–293

Singh AK (2011) Elemental chemistry and geochemical portioning of heavy metals in road dust from Dhanbad and Bokaro regions, India. Environ Earth Sci 62:1447–1459. doi:10.1007/s12665-010-0630-3

Sipos P, Kovacs Kis V, Marton E, Nemeth T, Zoltan M, Szalai Z (2012) Lead and zinc in the suspended particulate matter and settled dust in Budapest, Hungary. Eur Chem Bull 11:449–454

Steiner M, Bowler M, Schulz T, Pronk W (2007) Modelling heavy metal fluxes from traffic into the environment. J Environ Monit 9:847–854. doi:10.1039/B703509H

Swartjes FA (2011) Dealing with contaminated sites: from theory towards practical application, 1st edn. Springer, Berlin, p 264

Tanner PA, Ma H-L, Yu PKN (2008) Fingerprinting metals in urban street dust of Beijing, Shanghai and Hong Kong. Environ Sci Technol 42:7111–7117. doi:10.1021/es8007613

Tokalioglu S, Kartal S (2006) Multivariate analysis of the data and speciation of heavy metals in street dust samples from the organised industrial districts, Kayseri, Turkey. Atmos Environ 40:2797–2805. doi:10.1016/j.atmosenv.2006.01.019

Turner A (2011) Oral bioaccessibility of trace metals in household dust: a review. Environ Geochem Health 33:331–341. doi:10.1007/s10653-011-9386-2

Turner A, Hefzi B (2010) Levels and bioaccessibility of metals in dusts from an arid environment. Water Air Soil Pollut 210:483–491. doi:10.1007/s10653-011-9386-2

U.S. EPA (2008) Child specific exposure factors handbook (Final Report) 2008. U.S. Environmental Protection Agency, Washington, DC, EPA/600/R-06/096F

Wei B, Yang L (2010) A review of heavy metal contamination in urban soils, urban road dusts and agricultural soil from China. Microchem J 94:99–107. doi:10.1016/j.microc.2009.09.014

Wei B, Jiang F, Li X, Mu S (2009) Spatial distribution and contamination assessment of heavy metals in urban road dusts from Urumqi, NW China. Microchem J 93:147–152. doi:10.1016/j.microc.2009.06.001

Wei B, Jiang F, Li X, Mu S (2010) Contamination levels assessment of potential toxic metals in road dust deposited in different types of urban environment. Environ Earth Sci 61:1187–1196. doi:10.1007/s12665-009-0441-6

Wragg J, Cave M, Taylor H, Basta N, Brandon E, Casteel S, Gron C, Oomen A, Van De Wiele T (2009). Inter-laboratory trial of unified bioaccessibility procedure. British Geological Survey Open report, OR/07/027

Yongming H, Peixuana D, Junjib C, Posmentier ES (2006) Multivariate analysis of heavy metal contamination in urban dusts of X'ian, central China. Sci Total Environ 355:176–186. doi:10.1016/j.scitotenv.2005.02.026

Zhang M, Wang H (2009) Concentrations and chemical forms of potentially toxic metals in road-deposited sediments from different zones of Hangzhou, China. J Environ Sci 21:625–631. doi:10.1016/S1001-0742(08)62317-7

Zhao P, Feng Y, Zhu Y, Wu J (2006) Characterization of resuspended dust in six cities of North China. Atmos Environ 40:5807–5814. doi:10.1016/j.atmosenv.2006.05.026