

Characteristic comparison of heavy metal contamination between road-deposited and roof-deposited sediments in suburban area

Jianghua Yu^{1,2} · Haixia Yu³ · Liqiang Xu⁴ · Tao Tao^{1,2} · Yanxing Zhang^{1,2} · Yuhao Hua^{1,2} · Namwiinga Babra Nkolola^{1,2}

Received: 30 August 2016 / Accepted: 20 March 2017 / Published online: 1 April 2017
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Abstract Surface-deposited sediment in urban area is an essential environmental medium for assessing heavy metal contamination. A total of 10 sampling trips were conducted to collect road-deposited and roof-deposited sediments for the comparison of nickel (Ni), copper (Cu), cadmium (Cd), lead (Pb), zinc (Zn), and chromium (Cr) contamination characteristics. Results indicated that roof sediment appeared to have a finer size distribution than road sediments. Roof sediment indicated higher metal concentration and lower surface loading than road sediment. The impact of particle size on heavy metal contamination was quantified by using the developed pioneering term of finer particle effect factor; it differed according to surface types and grain size fraction. Particles in individual grain size fraction showed different contribution to the surface loading for bulk sediments. No consistent trend was found for the grain size fraction loading along with grain size for the studied heavy metals for road sediments. In

contrast, an asymmetric “W” trend was observed for the roof sediments, and it had the following results: Fraction of <63 and 250–500 μm showed higher loading, while fraction of 90–125 and >850 μm indicated the smaller loading. Findings above facilitated the appropriate management practice selection for the treatment of surface-deposited sediments.

Keywords Road and roof sediments · Heavy metals · Concentration · Size fraction loading

Introduction

As a special type of potentially toxic environmental medium with complicated composition in urban area, surface sediment (e.g., road sediment and roof sediment) contains a number of pollutants, such as heavy metals, nutrients, and hydrocarbons originated from a wide range of non-point sources including atmospheric deposition, vehicle exhausts, abrasion, and degradation of construction materials (Rijkenberg and Depree 2010; Shi et al. 2011; Zhao et al. 2014; Bian et al. 2015; Charters et al. 2016). It plays a major role in delivering chemical constituents during storms as they are transported to receiving water bodies (Kayhanian et al. 2012; Nguyen et al. 2014). Heavy metals associated with surface sediment are of primary concern because of their toxicity and non-degradability in the environment; their influence may be temporary, seasonal, or chronic (Kong et al. 2011; Fernández-Olmo et al. 2014). And the information on the contamination level of heavy metals can be considered as a valuable indicator of environmental pollution (Fujiwara et al. 2011). Thus, a great number of studies on surface sediment (especially road sediments) have focused on trace metal element concentration, distribution, and source identification in the last decades (Lu et al. 2009; Zhao et al. 2011; Li et al. 2015).

Responsible editor: Philippe Garrigues

✉ Haixia Yu
yuhaxia@mail.sysu.edu.cn

¹ Jiangsu Collaborative Innovation Center of Atmospheric Environment and Equipment Technology, School of Environmental Science and Engineering, Nanjing University of Information Science and Technology, Nanjing 210044, China

² Jiangsu Key Laboratory of Atmospheric Environment Monitoring and Pollution Control, Jiangsu Joint Laboratory of Atmospheric Pollution Control, Nanjing 210044, China

³ Guangdong Key Laboratory for Urbanization and Geo-Simulation, School of Geography and Planning, Sun Yat-sen University, Guangzhou 510275, China

⁴ Pearl River Hydraulic Research Institute of PRWRC, Guangzhou 510611, China

Many factors including quantity and grain size distribution can affect the environmental behavior of surface sediments. It is well recognized that finer sediments tend to have higher metal concentration and more easily be washed off into water bodies (Zhao et al. 2010; Zhao and Li 2013). It has been summarized that around 55–90% of particles are less than 125 μm in urban road runoff, and the corresponding value was 55–85% for roof runoff (Charters et al. 2016). Therefore, pollutants associated with these fine particles can be carried into water environment and cause a variety of environment problems (Bian et al. 2015).

Furthermore, due to the differences in pollution sources, road sediment is different from roof sediment. Hence, different contamination characteristics can be expected between road and roof sediments (Shi et al. 2011; Khanal et al. 2014; Yu et al. 2014). Shen et al. (2016) studied the particle size distribution and heavy metal contamination of three typical urban impervious surfaces in Beijing; results indicated that the cumulative percentage of grain size was different with this order roads in residential area > roofs > main traffic roads. It was also indicated that the heavy metal (lead, zinc, and manganese) concentrations were higher on the roof-deposited sediments than road sediments (Shen et al. 2016). The respective heavy metal contamination mean concentration of lead in cities was 240 mg/kg in Guangzhou city, China (Lu et al. 2009), 39.05 mg/kg in Ottawa city, Canada (Rasmussen et al. 2001), 120 mg/kg in Hong Kong (Yeung et al. 2003), and 236 mg/kg in Amman city, Jordan (Al-Khashman 2007). Additionally, the environmental and health effects of surface sediment are not only related to the pollutant concentration, the grain size, and the sediment quantity but also to the mobility of sediments in rainfall runoff and are further influenced by the pollution loading within individual grain size fraction (Sutherland et al. 2012; Zhang et al. 2016).

However, to our knowledge, very few studies have considered surface-deposited sediment contamination from a range of different surfaces in the same geographical area, although it is essential to have information on the comparison study for the effective management. Therefore, a deep understanding of heavy metal contamination characteristics in individual grain size for various kinds of surface sediment is crucial for assessment about the environmental behavior and fate of trace elements; this data is important for the model development for predicting loads from individual surface in a catchment for the selection of treatment facilities.

In this context, the primary objective of the present study is to compare the heavy metal contamination characteristics between road-deposited and roof-deposited sediments. The detailed focuses are to (i) clarify the difference of heavy metal concentration between road-deposited and roof-deposited sediments, (ii) determine and compare the effect of grain size on heavy metal contamination between road and roof sediments, (iii) reveal and compare the surface pollution loading for bulk

sediments and grain size fraction loading for size-dependent sediments between road and roof sediments, and (iv) give the treatment implications based on the contamination behavior.

Materials and methods

Site description

Samples were collected at Nanjing University of Information Science and Technology, which is located in the northern part of Nanjing City and is adjacent to the national Ningliu Road (see Fig. 1). The watershed around the sampling site (32° 12' 22.18" N, 118° 43' 08.07" E) is a typical suburban area with residential, agricultural, and industrial areas with steel plant, coal-burning power plant, and a chemical plant. Nanjing has humid subtropical climate influenced by the East Asian monsoon, with damp conditions throughout the year (the annual average relative humidity is around 76%). It is very hot in summer (June–August) with the average temperature of 27.7 °C in Nanjing; the annual average rainfall depth is around 1062 mm, and half of the total rainfall happens in summer.

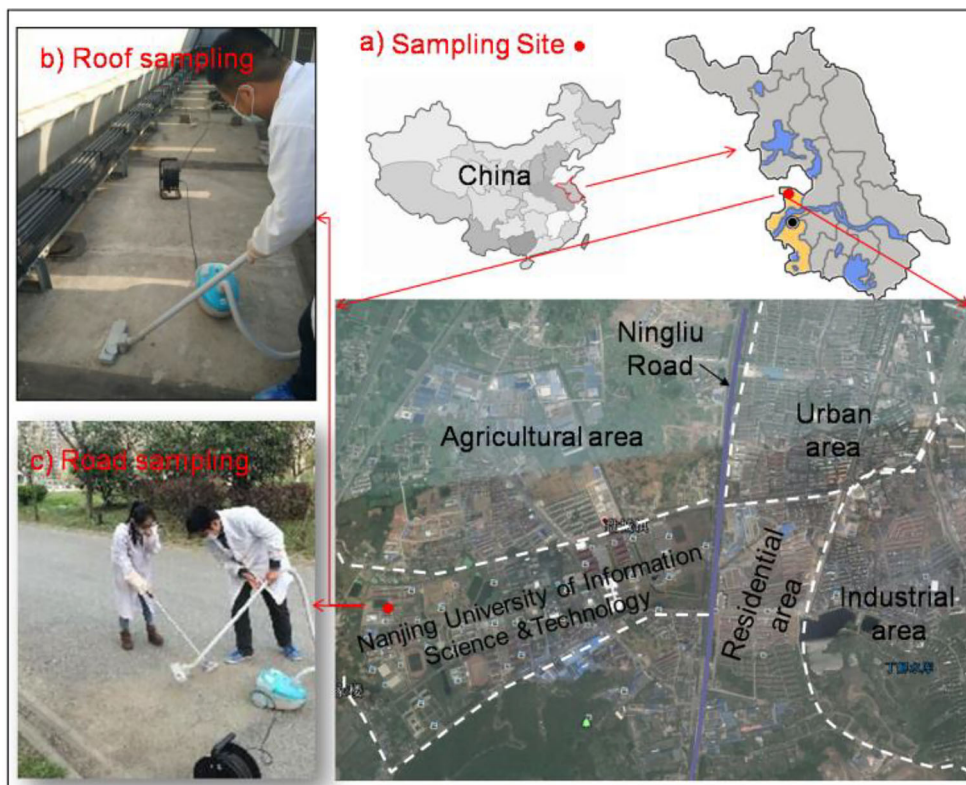
Sample collection and pretreatment

A total of 10 sampling trips were conducted from September 2014 to February 2016, and the sample information is listed in Table 1. The sampling roof is on the top of a 25-m high building with a total of five floors. The roof is mainly made of concrete material with some asphalt slim belt surrounding and some rusted cast iron pipes and other steel appurtenances installed above the roof top around 25 cm. The sampling road is typical asphalt which is widely constructed in China. The road-deposited and roof-deposited sediments were collected using a dry vacuuming system. The field sampling process is briefly summarized below.

Firstly, the selected sample plots were demarcated at each sampling site. Then, the demarcated area was lightly scrubbed using a fiber brush to dislodge most of the fine particles attached to the surface. A consistent light approach was used when scrubbing the surface to avoid breaking up the parent road surface material and to ensure a homogeneous sample of solids (vaze and Chiew 2002). Secondly, a dry sample was collected from the selected plot using a vacuum cleaner. This process was repeated three times to ensure that all the possible particles were collected. After that, all the collected sediments were mixed and transferred to a polyethylene container and taken back to the laboratory for analysis.

The collected surface samples were dried in natural conditions for 2 weeks and then dried to constant weight in an oven at 105 °C for 24 h. The prepared samples were size-fractionated by separating them using stainless steel sieves into six particle size ranges: <63, 63–90, 90–125, 125–250,

Fig. 1 a Sampling site. b Roof sediments. c Road sampling



250–500, 500–850, and >850 μm . Particulates in the above seven size ranges are typically referred as silt and clay (<63 μm), very fine sand (63–125 μm), fine sand (125–250 μm), medium sand (250–500 μm), and coarse sand (>500 μm) (Bian et al. 2015; Zhao et al. 2010). Then, the separated solid samples were stored in the sealed polyethylene bags and labeled for the measurement of heavy metals.

The sediment digestion can be briefly described as follows: A small portion (around 0.5–1.0 g) of the sediment sample in each

size range was digested around 1.5 h with aqua regia and HClO_4 at 120 $^\circ\text{C}$ on a electric hot plate and then at 250 $^\circ\text{C}$ evaporated to almost dryness; the remaining residues were dissolved in 100 ml of deionizer water and filtered with fiber membrane; the filtrate was acidified at pH 1 and kept in 100-ml volumetric flask for metal measurement. The concentration of nickel (Ni), copper (Cu), cadmium (Cd), lead (Pb), zinc (Zn), and chromium (Cr) was determined. The measurement procedure of metals was as the same in our previous study (Yu et al. 2015).

Table 1 Specific information of sampling

Dust sampling							
No.	Sampling date	Antecedent dry periods (days)	Sample area (m^2)		Sample quantity (g)		
			Road	Roof	Road	Roof	
1	17 September 2014	5	5.7	2	566	136	
2	04 November 2014	6	5.7	2	642	140	
3	21 December 2014	5	5.7	2	526	192	
4	06 March 2015	3	5.7	2	332	152	
5	12 April 2015	7	5.7	2	256	184	
6	24 May 2015	3	5.7	2	339	221	
7	13 September 2015	5	5.7	2	315	175	
8	18 October 2015	2	5.7	2	326	159	
9	29 November 2015	5	5.7	2	452	172	
10	21 February 2016	6	5.7	2	475	191	

Calculation methods

Heavy metal concentration of bulk sediments

The general concentration of metals for bulk sediments (C_{Bulk}) is calculated as follows (Zhang et al. 2016):

$$C_{\text{Bulk}} = \sum_{i=1}^7 C_{ij} \times F_{ij}$$

where C_{ij} and F_{ij} are the metal concentration (mg/kg) and sediment mass fraction, respectively, for size grading j of sampling i .

Finer particle effect factor

In this study, in order to highlight the size control on the heavy metal concentration, a term of finer particle effect factor (F_{FPE}) was applied. It was defined as the concentration ratio of heavy metals between sediments in individual size fraction and bulk sediments. Generally, while the value of F_{FPE} is greater than 1, it denotes that the sediments are pollutants enriched compare with the bulk sediments; otherwise, the sediments are pollution diluted. The below formula was developed:

$$F_{\text{FPE}} = \frac{C_{ij}}{C_{\text{Bulk}}}$$

Surface loading

The formula below was developed to calculate the surface loading of deposited sediments:

$$\text{Load}_s = \frac{1}{A_s} \sum_{i=1}^n \left(\frac{\sum_{j=1}^7 C_{ij} \times M_{ij}}{T_i} \right)$$

where A_s is the area within which sediments were collected (m^2), M_{ij} is collected sediment mass (kg), for size grading j of sampling i , and T_i is the antecedent dry period before sampling for sample i (h).

Grain size fraction loading

The grain size fraction loading (Load_{GSF}) is calculated to determine the pollution contribution of metals in size-dependent sediments (Zhao et al. 2010):

$$\text{Load}_{\text{GSF}} = \frac{C_{ij} \times M_{ij}}{\sum_{i=1}^7 C_{ij} \times M_{ij}} \times 100\%$$

Results and discussion

Grain size distribution

The grain size distribution of surface-deposited sediments is an essential factor of determining the particles' mobility in storm water runoff and the contamination of the associated pollutants (Li et al. 2015; Zhao et al. 2010). As shown in Fig. 2, it is clear that a big difference for grain size distribution was obtained between road-deposited and roof-deposited sediments. The value of D_{50} (the value of the particle diameter at 50% in the cumulative distribution) was in the range of 210–700 μm (average of 405 μm) and 150–295 μm (average of 212 μm), respectively, for road-deposited and roof-deposited sediments. Result of independent-sample t test indicated that D_{50} of road sediments was significantly different from roof sediments with the significance level of 0.001. For road-deposited sediments, most of the solids were in the range of 125–1000 μm , accounting for around 62% (average value) of the bulk sediments by weight. This was consistent with the trend reported in previous studies (Li et al. 2015). And it was another case for roof-deposited sediments, around 80% of the bulk sediments by weight distributed in the ranges of <90 and 125–500 μm . Especially, particles in the range of 250–500 μm accounted for around 31% of the bulk sediments. The difference in particle size distribution could be resulted by the constructed surface material, the sediment source, and the influence of human activities including traffic density and sweeping. The main sources of road-deposited sediments are vehicle exhausts, abrasion, degradation of road materials, and artificial processes like traffic and road cleaning activities, while the main sources of roof-deposited sediments are atmospheric deposition, degradation of construction materials, and natural processes which include wind erosion and rain erosion (Charters et al. 2016; Khanal et al. 2014).

Metal concentration

Generally, increases in metal concentration were obtained with decreasing grain size for both road-deposited and roof-deposited sediments with exception of Pb; an observation of fluctuation for Pb along with the grain size was found (see Fig. 3). Anyway, this trend was generally in agreement with the results observed in previous studies due to the fact that the finer particles have greater surface area per volume unit and higher cation exchange capacity, which, in turn, indicate higher adsorption capacity for heavy metals (Bian et al. 2015; Li et al. 2015). Also, it appeared that the element pollution may be ranked as $\text{Zn} \gg \text{Cu} > \text{Cr} > \text{Ni} > \text{Pb} \gg \text{Cd}$ for both road-deposited and roof-deposited sediments. More importantly, heavy metal concentrations were found to be significantly different ($P < 0.01$ for all six studied heavy metals) from each other for road-deposited and roof-deposited

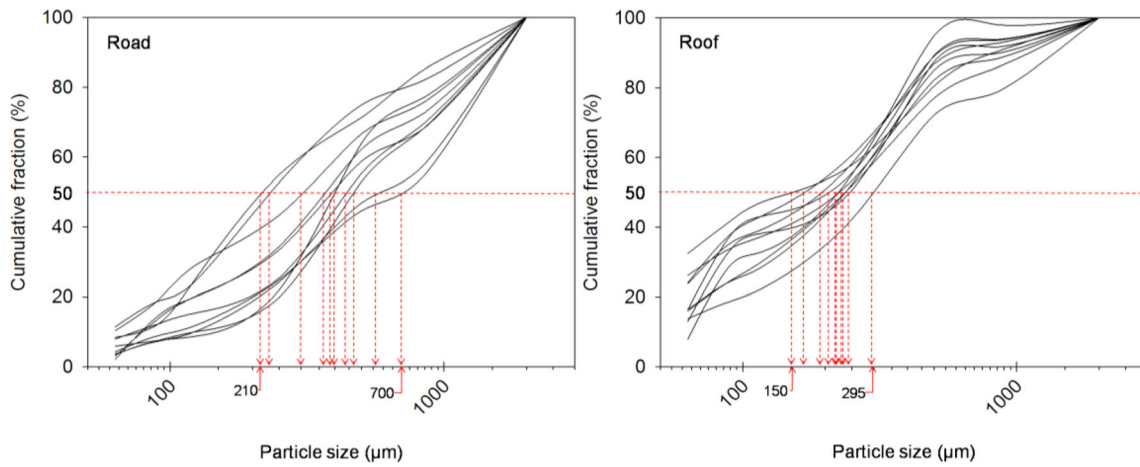


Fig. 2 Particle size distribution comparisons between road and roof sediments

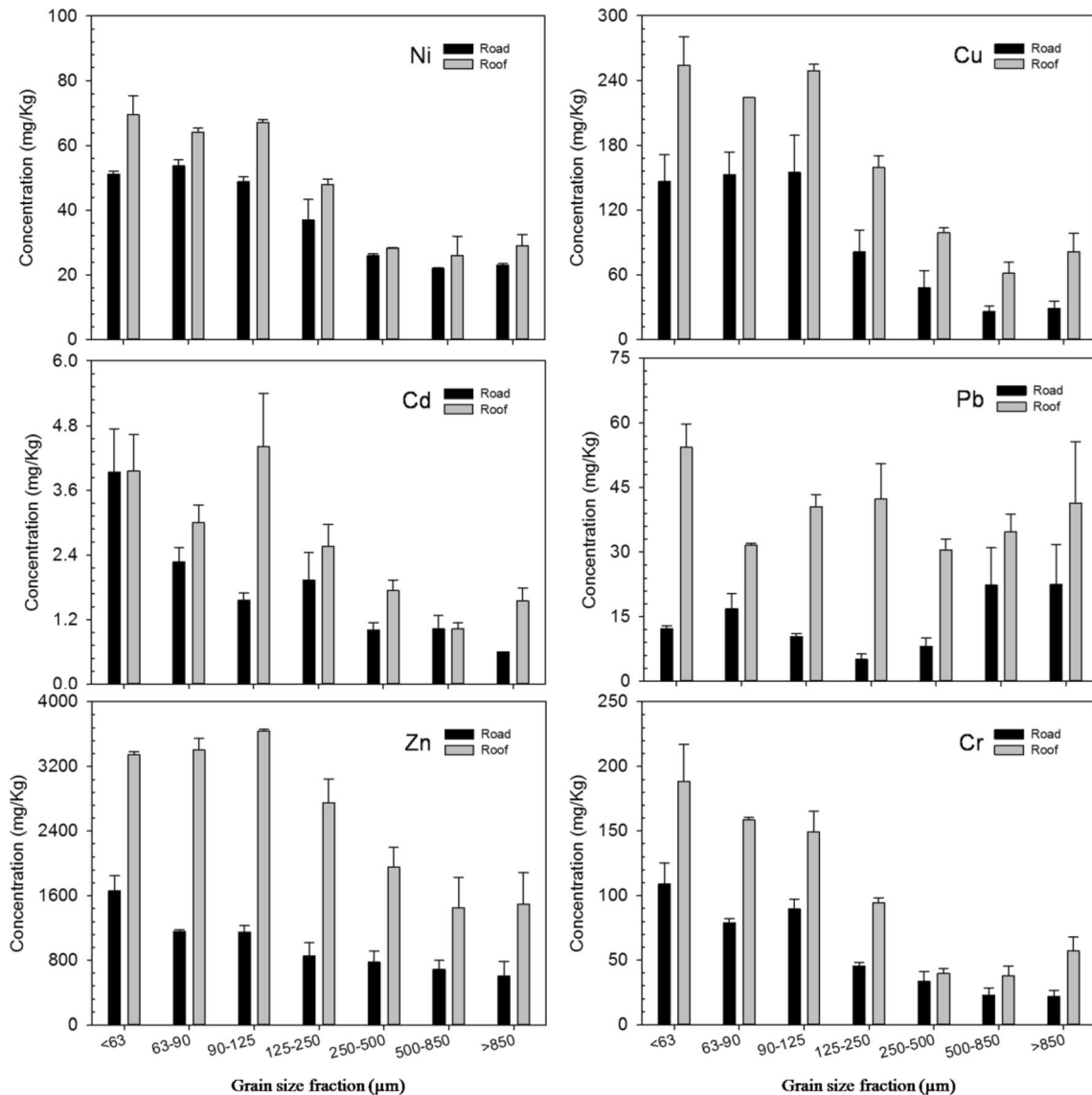


Fig. 3 Metal concentration comparison between road and roof fraction sediments

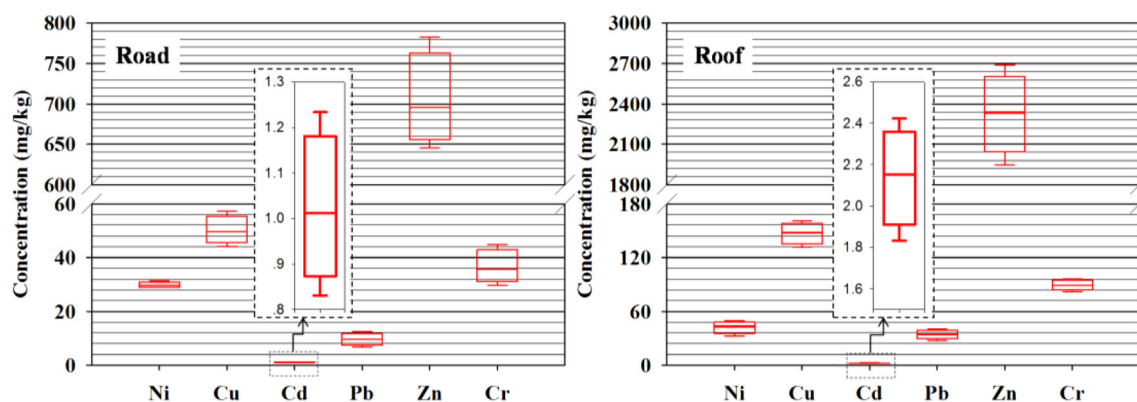


Fig. 4 Metal concentration comparison between road and roof bulk sediments

sediments in individual grain size. Metal concentration of roof sediments was higher than that of road sediments, and this difference was particularly obvious for Zn, Pb, and Cr. This difference was further supported by the metal concentration of bulk sediments (see Fig. 4). The respective average concentration of Ni, Cu, Cd, Pb, Zn, and Cr for road-deposited sediments was 29.9 ± 1.0 (mean \pm standard error of the mean), 50.2 ± 5.4 , 1.1 ± 0.1 , 9.6 ± 2.4 , 704.9 ± 57.4 , and 36.5 ± 6.3 mg/kg. The corresponding value for roof-deposited sediments was 43.5 ± 0.9 , 147.1 ± 4.0 , 2.1 ± 0.2 , 34.2 ± 1.6 , 2329.9 ± 222.9 , and 89.0 ± 1.3 mg/kg. It was reported that the concentration value in soil of China was 26.9, 22.6, 0.097, 26, 100, and 61 mg/kg, respectively, for Ni, Cu, Cd, Pb, Zn, and Cr (Wei and Yang 2010). Therefore, it can be stated that the pollution concentration of the studied six heavy metals is different. Especially, concentration of Pb for both road and roof sediments in this study was evidently lower than the concentration value in the soil. It can be deduced that Pb contamination in this study is mainly caused by human activities (e.g., waste of tyre and brake pad) other than soil erosion.

It is known that the roof-deposited sediment is mainly from the atmospheric deposition and the compounds leaching from roof materials (Charters et al. 2016; Yu et al. 2014), while the road-deposited sediment is a multiple complicated source of traffic wastes (tires, brakes, and emission), atmospheric deposition, erosion of surrounding soils, and road material wear (Zhao and Li 2013; Khanal et al. 2014). Specially, the road surface has substantial sediment inputs from vehicle component wear and wash-off from car bodies that are absent from roof sediments (Charters et al. 2016). Thus, the composition of deposited sediment is different for various kinds of surface within the same catchment, which, in turn, results in the difference in the nature of the sediments. In addition, the mineral composition of sediments from different kinds of surface could be different (Yu et al. 2015). Discussion above explains at some degree why there is a sharp difference for the heavy metal contamination between road and roof sediments. However, the higher metal concentration of roof-deposited

sediments does not definitely mean greater pollution loading to environment system considering the difference of bulk sediment quantity between roof and road sediments.

Finer particle effect factor

It has been reported frequently that the pollution of surface-deposited sediment is size control (Zhao et al. 2011; Yu et al. 2015), while very few studies quantify the specific impact of grain size on sediment contamination. Thus, the term of finer particle effect factor which highlights the phenomenon of finer particles and higher heavy metal concentration was applied, and the results are shown in Fig. 5. Generally, the studied heavy metals except Pb showed notable finer particle effect for both road and roof sediments. This is consistent with the results in most published references that chemical concentration has been found to be higher in fraction of fine particles (Khanal et al. 2014). And this is in agreement with the results shown in Fig. 3. The exception of Pb could be caused by the pollution source as discussed above. The seven size ranges can be categorized into three groups based on the values of finer particle effect factor. The ratio was larger than 1 evidently for particles less than $125 \mu\text{m}$, and it was around 1 for particulates in the range of $125\text{--}250 \mu\text{m}$, while it was evidently lower than 1 for solids coarser than $250 \mu\text{m}$. More interestingly, it should be pointed out that this ratio was more close to 1 for particles coarser than $250 \mu\text{m}$ of road sediments than roof sediments in the same size range. This is probably caused by the fact that the heavy metal contamination of road sediment is more frequently influenced by the traffic activities, and the metals in the coarse particles are mainly derived from the components of wear and tear of vehicles (Li et al. 2015). Hence, we can conclude that the sediments less than $125 \mu\text{m}$ contain a greater proportion of the pollutants than the mass percentage of sediments in this size range. And this was verified by the data list in Table 2. It is clear that the heavy metals' mass percentages were greater than the corresponding values of sediments less than $125 \mu\text{m}$ for both road-deposited and roof-deposited sediments. Based on the average values,

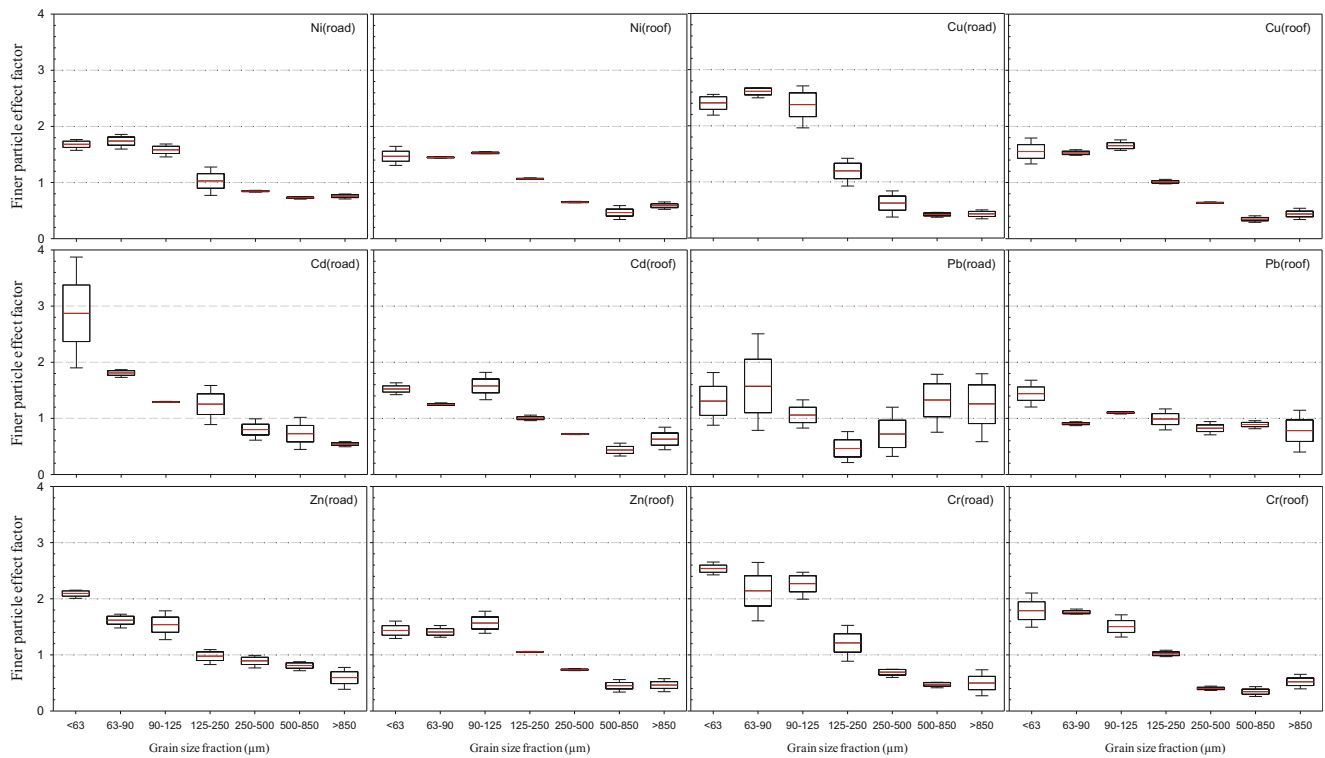


Fig. 5 Finer particle effects of heavy metals along with grain size (*red line* indicates the mean value)

around 34.5% of Ni, 50.5% of Cu, 37.3% of Cd, 26.9% of Pb, 37.2% of Zn, and 44.8% of Cr were associated with around 21.4% of solids less than 125 μm for road sediments, and around 53.2% of Ni, 56.6% of Cu, 51.5% of Cd, 44.4% of Pb, 53.5% of Zn, and 62.9% of Cr were associated with around 37.2% of solids less than 125 μm for roof sediments.

In addition, it can be found that road sediments showed a more apparent finer particle effect than roof sediments. This is to say that the heavy metal contamination of roof-

deposited sediment was more homogeneous in regard of grain size control. This difference may be resulted by the diversity of metal source, grain size distribution, and human activities (e.g., traffic, land use) between road and roof sediments. Based on the findings above, we can say that the surface sediments in the finer size fraction (e.g., less than 125 μm) is important due to the fact that these sediments are higher contaminated and are preferentially transported during storm runoff process.

Table 2 Mass percentages of metals associated with solids less than 125 μm

No.	Road							Roof						
	Sediments	Ni	Cu	Cd	Pb	Zn	Cr	Sediments	Ni	Cu	Cd	Pb	Zn	Cr
1	12.8	22.9	33.9	27.4	25.9	20.2	27.1	33.6	53.4	58.5	53.0	37.5	54.2	63.5
2	31.3	48.1	69.3	52.2	25.9	57.2	62.1	37.9	52.9	54.3	50.8	47.6	49.8	63.3
3	17.7	30.4	45.7	34.9	22.6	31.2	41.4	38.5	56.0	59.2	54.8	46.5	54.6	65.9
4	20.6	33.8	49.6	38.2	24.8	36.2	43.5	36.6	54.1	57.3	52.9	43.9	52.8	64.2
5	22.0	36.4	50.1	39.8	22.0	41.3	43.2	37.2	51.5	55.8	49.2	41.9	56.0	61.5
6	19.0	31.0	46.8	36.2	20.5	38.5	41.5	35.6	49.3	55.2	52.3	42.2	55.6	62.3
7	26.5	42.8	61.2	35.1	36.2	40.2	55.6	43.2	62.7	69.1	55.6	52.3	56.1	71.4
8	19.8	32.5	45.2	33.3	31.3	29.8	43.2	36.0	53.2	55.2	51.5	42.1	48.2	59.2
9	23.9	36.6	55.3	41.1	25.7	43.2	51.8	35.8	49.0	53.4	48.5	42.3	53.3	60.1
10	20.8	31.7	48.9	36.4	27.6	36.5	41.7	37.6	51.8	50.5	48.6	46.2	55.0	59.5
Average	21.4	34.5	50.5	37.3	26.9	37.2	44.8	37.2	53.2	56.6	51.5	44.4	53.5	62.9
SD	5.0	7.0	9.6	6.4	4.6	9.7	9.4	2.6	3.8	5.1	2.6	4.1	2.7	3.7

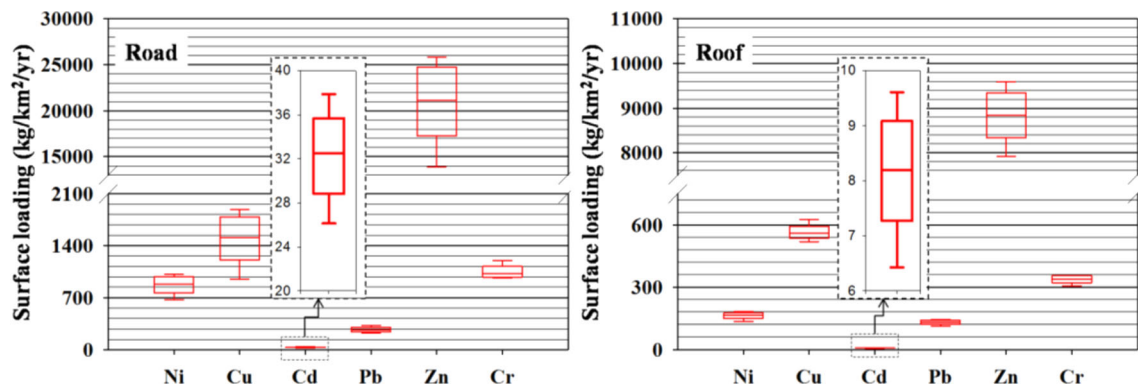


Fig. 6 Metal surface loading comparison between road and roof sediments

Pollution loading

Surface loading of bulk sediments

In order to determine the heavy metal pollution loading of surface-deposited sediments, the surface loading was calculated based on bulk sediments, and the results are shown in Fig. 6. Overall, the road-deposited sediments showed higher surface loading for the six studied metals than roof sediments based on the bulk sediments. This was opposite to the comparison results of heavy metal concentration as shown in Fig. 3. The reason should be the fact that the sediment loading of road was higher than that of roof. The respective mean values of surface loading of Ni, Cu, Cd, Pb, Zn, and Cr for road-deposited sediments were 874.0 ± 128.8 (mean \pm standard error of the mean), 1502.2 ± 350.2 , 32.3 ± 4.2 , 270.0 ± 37.0 , $21,004.3 \pm 4507.2$, and 1042.1 ± 92.2 kg/km²/year. The corresponding value for roof-deposited sediments was 165.1 ± 6.4 , 558.3 ± 20.4 , 8.2 ± 1.1 , 130.4 ± 10.9 , 8814.6 ± 445.5 , and 338.4 ± 20.3 kg/km²/year. Thus, it can be concluded that the road-deposited sediments indicated a higher pollution loading to the environment system than roof-deposited sediments within the same catchment area; even the roof sediments showed a higher metal concentration level.

Grain size fraction loading

The total surface loading is a summation of pollution loading caused by the solids in individual size fraction; an important index of contamination for surface-deposited sediments is the mass load of heavy metals in a given grain size fraction (Li et al. 2015; Zhu et al. 2008). In order to investigate the loading contribution of particulates in individual size fractions, the grain size fraction loading was calculated, respectively, for road and roof sediments, and the results are included in Fig. 7.

Overall, grain size fraction loading indicated different distribution pattern in various size fraction for road and roof sediments. In terms of road-deposited sediments, no evident

trend was found for grain size fraction loading along with grain size. This may be due to the fact that the grain size fraction loading was a combination of concentration and size distribution, and the size distribution was different for road and roof sediments (see Fig. 2). And solids in the range of 250–500 μm showed a relatively higher loading fraction than solids in other size fractions; the respective average value in this size fraction was 24.0 ± 5.3 , 18.3 ± 9.1 , 23.1 ± 8.8 , 21.4 ± 14.8 , 25.6 ± 7.7 , and $19.5 \pm 4.0\%$, respectively, for Ni, Cu, Cd, Pb, Zn, and Cr for road-deposited sediments. This relative higher pollution loading was mainly caused by the higher solid mass percentage (average of 28%) in the size range. Further, the solids in the range of 500–850 μm showed the lowest loading fraction for the studied heavy metals with exception of Pb; the average loading fraction was no more than 9% for the other five studied heavy metals. The distribution pattern of size fraction loading for road sediments in our study was different with the result obtained in published reference. Zhu et al. determined the size fraction loading of five metals (Zn, Pb, Cu, Cr, and Ni) for road-deposited sediments with various land use; results indicated that the distribution pattern of size fraction loading varied with different land use, and a general decreasing trend was found along with increasing grain size (Zhu et al. 2008). Normally, coarser particles, e.g., particles in the range of 250–500 μm , would be less likely to appear in the receiving waters due to the fact that they could be effectively removed by the conventional road sweeping (Chang et al. 2005; Herngren et al. 2006). This further strengthens the importance of effective sweeping activities to remove the particles within the range of 250–500 μm from road-deposited sediments for the pollution mitigation to aquatic environment.

It was another case for roof-deposited sediments. The value of surface loading indicated asymmetric “W” trend along with grain size fraction for all the six studied heavy metals as shown in Fig. 7. Solids in the range of 90–125 and 500–850 μm indicated the lowest surface loading for the studied heavy metals with the fraction of 5.0–8.9 and 1.8–5.1%, respectively. The peak values of surface loading were

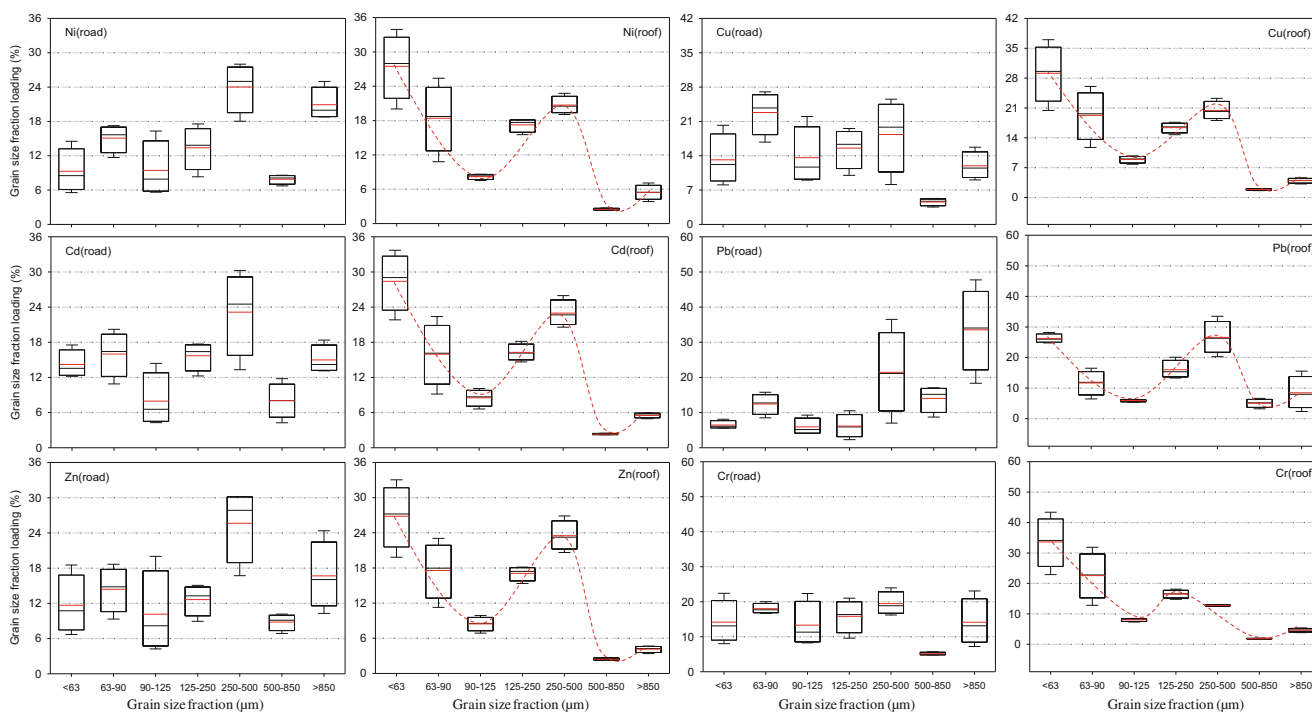


Fig. 7 Metal grain size fraction loading comparison between road and roof sediments (red line indicates the mean value)

contributed by the solids in the range of <63 and 250–500 μm with the contribution of 26.3–33.6 and 20.4–26.6%, respectively, which accounted for around 50% pollution loading for the studied heavy metals. Due to the fact that the average mass percentage of particles in the range of <63 and 250–500 μm was 19 and 31%, respectively (see Fig. 2), it can be deduced here that the higher pollution loading was mainly resulted by the higher heavy metal concentration for particles less than 63 μm, and it was mainly caused by the higher solid mass fraction for particles in the range of 250–500 μm.

On the other hand, it is well known that the mobility of surface sediments in runoff mainly depends on the grain size, and the finer solids could be easier washed off and brought into receiving waters (Bian et al. 2015; Zhao et al. 2010). Thus, we can say here that the higher grain size fraction loading does not definitely lead to higher pollution potential to receiving waters. Pollution loading in various size fractions should have different potential to receiving waters. To a certain extent, the heavy metal pollution potential of urban surface sediments to receiving waters is determined by the contamination of fine particles which could be mobilized in runoff (Zhang et al. 2016).

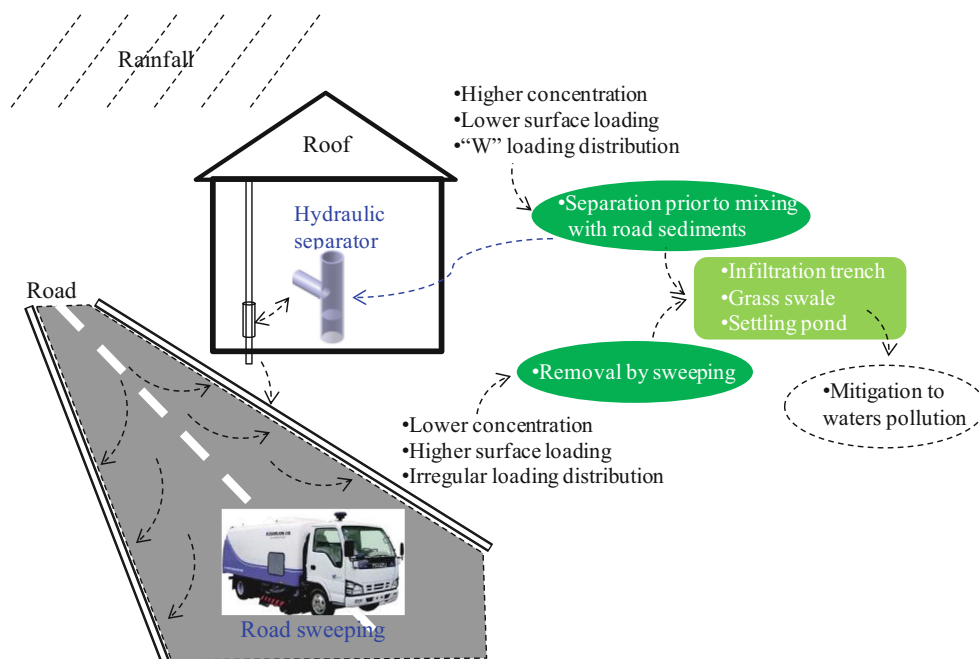
Implications for treatment approaches

Overall, different types of surface-deposited sediment showed variant contamination characteristics. Road-deposited sediment has lower heavy metal concentration while indicated higher surface loading compared with roof-deposited

sediments. Sediments in different size fraction have different pollution loading to total pollution of bulk sediments, and the distribution model of pollution loading along with grain size was different for road and roof sediments. For road-deposited sediments, solids in the range of 250–500 μm dominated the pollution loading of bulk sediments. Due to the fact that solids equal to or bigger than this size could be easily collected through conventional street sweeping activities (Chang et al. 2005; Hengren et al. 2006), the selection of an appropriate sweeping technology becomes more important for the mitigation of pollution load to environment (see Fig. 8). As to the treatment of particles less than 250 μm, some practices such as filtration trench, grass swale, and settling pond can be recommended to install prior to the storm water network for the pollution mitigation to water environment.

In terms of roof sediments, considering the fact that the roof sediments would be washed off into storm water discharge network through roof downpipe, and this runoff carried with roof sediments would be mixed in the kerb and channel with road runoff which transported with road sediments. Therefore, the sediments carried in roof runoff becomes concentration diluted for associated pollutants; even the runoff waters would be normally pollution elevated, and the combined runoff would require a larger capacity treatment system due to the greater mixed volume. And the degree to which it is diluted depends on the composition (ratio of roof area to road area) of the watershed that the kerb and the channel serve (Charters et al. 2016). In this context, separation of sediments, especially the particles with higher pollution loading, from roof runoff

Fig. 8 Effective treatment practices for surface-deposited sediments



prior to it mixing in the kerb and channel may be a more effective approach for the treatment of sediments and runoff. Facility such as hydraulic separator (e.g., hydrocyclone and vortex separator as shown in Fig. 8) is optimal and recommended to install at the bottom point of roof downpipe to effectively remove sediments (Yu et al. 2013).

Conclusions

The characteristic comparison of heavy metal pollution between road-deposited and roof-deposited sediments was conducted in the present study. The following important conclusions were derived from this research.

A significant difference in heavy metal concentration was observed between road-deposited and roof-deposited sediment. Roof sediment showed a higher concentration level and lower surface loading than road sediments. Contamination of surface-deposited sediment was size-dependent. Specially, particles less than 125 μm were heavy metals enriched compared with the bulk sediments for both road and roof sediments. Particles in different size range showed different grain size fraction loading for the studied heavy metals for surface-deposited sediments. And the distribution of grain size fraction loading within the various size ranges for heavy metals was also different for road sediments and roof sediments. Pollution loading in the grain size of 250–500 μm was dominant for road sediments and accounted for about 20% for the studied six heavy metals. For roof sediments, <63 and 250–500 μm fractions were dominant with around 50% pollution loading for the studied six heavy metals.

The implications of the findings obtained in the present study are that a comprehensive consideration should be taken into account for the treatment of road and roof surface sediments. The particle size distribution, heavy metal concentration, and grain size fraction loading are essential factors affecting the selection of an appropriate management practice for surface sediments. Surface-deposited sediments with different pollution features should be treated with individual treatment approach.

Acknowledgements The data was provided by the National Natural Science Foundation of China (grant number 41401546), and the preparation of this paper was funded by the National Natural Science Foundation of China [grant numbers 41301627 and 21501097] and Water Science Innovation Program of Guangdong, China [grant number 2014-12].

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