1   **Risk Assessment of Exposure to Particulate Output of a Demolition Site**

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6   **i.   Abstract**

7   Whilst vehicular and industrial contributions to the airborne particulate budget are well explored, the input due to
8   building demolition is relatively unknown. Air quality is of importance to human health and it is well known that
9   composition of airborne particles can have a significant influence on both chronic and acute health effects. Road
10   dust (RD) was collected before and after the demolition of a large building to elucidate changes in elemental profile.
11   Rainfall and PM_{10} mass concentration data aided interpretation of the elemental data. Quantification of Al, As, Ba,
12   Ca, Cd, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, Pb, Rh, S, Si, Sn, Ti, V and Zn was carried out. It was found that only Al, K, Mg,
13   Si and S increased in concentration across all size fractions after the building demolition. Risk assessment was then
14   carried out on elements with applicable reference dose values to assess the potential health risks due to the
15   demolition. Significant risk to children was observed for chromium and aluminium exposure. PM_{10}, monitored 40
16   meters from the demolition site, indicated no abnormal concentrations during the demolition, however, rainfall data
17   was shown to affect the concentration of PM_{10}. The elemental data observed in this study could possibly indicate
18   the role of increased sulphur concentrations (in this case as a result of the demolition) on the buffer capacity of RD,
19   hence leaching metals into rainwater.

20   **ii.   Keywords**

21   Road Dust, Demolition, Elemental Concentration, Buffer Capacity

22   *1.   Introduction*

23   The contribution to local particulate matter (PM) concentrations originating from building construction and
24   demolition is seldom acknowledged and not considered significant to the PM budget. Literature on the topic is
25   sparse. A study by Deacon et al. (1997) touched upon the possible contribution of a building demolition to mass
26   concentration of PM less than 10µm (PM_{10}) in the early years of Automatic Urban Air Quality Monitoring Network
27   (AUN), UK. Results indicated that daily maximum exceedances of PM_{10} had doubled in a period when demolition of
28   a building had been carried out near a monitoring station in Cardiff, UK. With the exception of a study by Liu et al.
29   (2014) whom only attributed changes in calcium and barium concentrations to construction/demolition work, there
30   is a corresponding paucity in the literature on changes of metal concentrations due to building work. Due to the
31   recognised fact that atmospheric PM poses a threat to human health (for example increased mortality amongst the
32   general population as an effect of exposure to PM (Pope et al. 2002; Villeneuve et al. 2002; Meister et al. 2012;
33   Sorenson et al. 2012), the contribution to the PM budget during and after demolition episodes are a concern. In fact,
34   the World Health Organisation suggests that particulate matter is responsible for 8% of lung cancer deaths, 5% of
35   cardiopulmonary deaths and 3% of respiratory infection deaths (WHO 2009). Although vehicular emissions are most
36   commonly cited as a key source of particles (Gunawardana et al. 2012; Dong and Lee 2009 and references therein),
37   some acknowledgement to the role of industrial processes are given. These health effects are still mainly focussed
38   on the PM mass concentrations and certain aspects regarding the composition and generation of particles remain
39   unaddressed and unaccounted for.

40   For the purpose of this study, particles of specific interest are those which gather as curb-side sediment ready to be
41   pulverised and suspended by the actions of meteorological conditions or vehicles making use of the road (Abu-
42   Allaban et al. 2003; Atiemo et al. 2012). These particles may be referred to as road dust (RD). Research indicates that
43   road dust accounts for up to 74% of total suspended particulate matter (TSP) in Ho Chi Minh City, Vietnam (Hien et
44   al. 1999) and 62% in Lahore, Pakistan (Harrison et al. 1997). This resuspension of RD can enable it to become respired
45   by pedestrians and other road users. The most appealing aspect of RD from an analytical standpoint is the already
46   well-documented abundance of transition metals, many of which are potentially toxic and in some incidences

carcinogenic (Unceta et al. 2010). Often the effects manifest themselves as cardiorespiratory problems, both chronic and acute, see table 1.

The intention of this study is to investigate the variation in concentration of elemental components of RD before and after the demolition of a large building at an urban site. The selected elements are: Al, As, Ba, Ca, Cd, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, Pb, Rh, Si, Sn, S, Ti, V and Zn. Analysis of these elements is justified by their potential toxicity to humans during overexposure. This research will focus on the resuspendable fraction of RD, which for the purpose this study is defined as particles up to around 140μm in accordance with the research by Kennedy and Hines (2002). The elemental concentrations will be subject to risk assessment to decipher the possible effects on human health. The risk assessment method used in this paper is based on the United States Environmental Protection Agency (USEPA) model (USEPA, 1997), modified for use with sediments as demonstrated on previous studies involving the risks associated with RD (Ma and Singhirunnusorn 2012; Du et al. 2013). This will elucidate whether the anticipated increase in metal concentrations are enough to cause significant increase to likelihood of developing detrimental health effects because of contaminant exposure.

Table 1. Common metal contaminants of RD and their potential effects on human health

Metal	Health concerns	Citation
Al	Neuropathological, neurophysical and neurochemical changes	Miu et al. 2003
As	Carcinogenic	Zhang et al. 2014
Ba	Hypertension	Oskarsson 2015
Cd	Carcinogenic	Zhang et al. 2009
Cr	Carcinogenic in +6 state	Goldbohm et al. 2006
Cu	Free radical production	Kadiiska and Mason 2002
Fe	Causes eye problems	Ugarte et al. 2013
Mn	Heart conditions	Cavallari et al. 2008
Ni	Carcinogenic	Kasprzak et al. 2003
Pb	Affects brain and nervous system	Zhang et al. 2013
Rh	Cytotoxicity	Iavicoli and Leso, 2015
Si	Causes silicosis	Naghadehi et al. 2014
Sn	Gastrointestinal conditions	Ostrakhovitch 2015
V	Affects nervous system	Li et al. 2013
Zn	Pulmonary effects	Mueller and Seger 1985

2. Experimental Methodology

### 2.1 Sample Site

The RD sample site is a traffic island (British National Grid reference: SJ 84229734) in the meridian of Oxford Road, Manchester, UK (Figure 1). Oxford Road is one of the most highly trafficked roads in Manchester supporting traffic from surrounding settlements of Greater Manchester, with buses and cars being the predominant mode of transport. Another feature present at this section of Oxford Road is the raised train tracks running perpendicular between two of the main train stations in Manchester. Aside from these vehicles, the roadside area is heavily used by pedestrians, with a large cohort being the student population of the local universities, supporting approximately 75,000 students. The sampling site is situated 20 m west of the 16600 m^2 demolition site and approximately 40 m north of one of Manchester's air monitoring station. The building subject to demolition was the 'New Broadcasting House', more commonly known as the BBC Building, Manchester, and constructed in the early 1970s (1971-1975). The building is mostly formed of steel reinforced concrete, with bronze-coated window panes and aluminium window frames. The soft demolition took place between November 2012 and May 2013. It was noted that throughout the process, water was used to minimize the spread of dust from the site to surrounding areas.

2.2 Sample collection and processing

Collection of road dust was undertaken at the onset and the completion of the building demolition (November 2012 and May 2013, respectively). Using a polyethylene dust pan and brush as recommended by Charlesworth and Lees (1999); and placed in polyethylene bags for transfer to the laboratory. Approximately 10 kilograms of sample mass was collected from across the site.

1   After collection, the RD was transferred to the laboratory to be air-dried in paper soil bags at 25 ± 5°C until a
2   consistent weight was achieved, between 14-21 days. Dry samples were fractionated into particles of four grain
3   sizes: <38μm, 63-38μm, 125-63μm and 2000-125μm, where these particles represent the boundaries between 'silts
4   and clays', 'course silts', 'very fine sands' and 'coarse material' respectively (Tucker 1991). Analysis is restricted to
5   the three finest fractions of RD as those are capable of being resuspended. The 2000-125μm fraction is retained for
6   further unreported analysis, the >2000μm fraction is discarded.
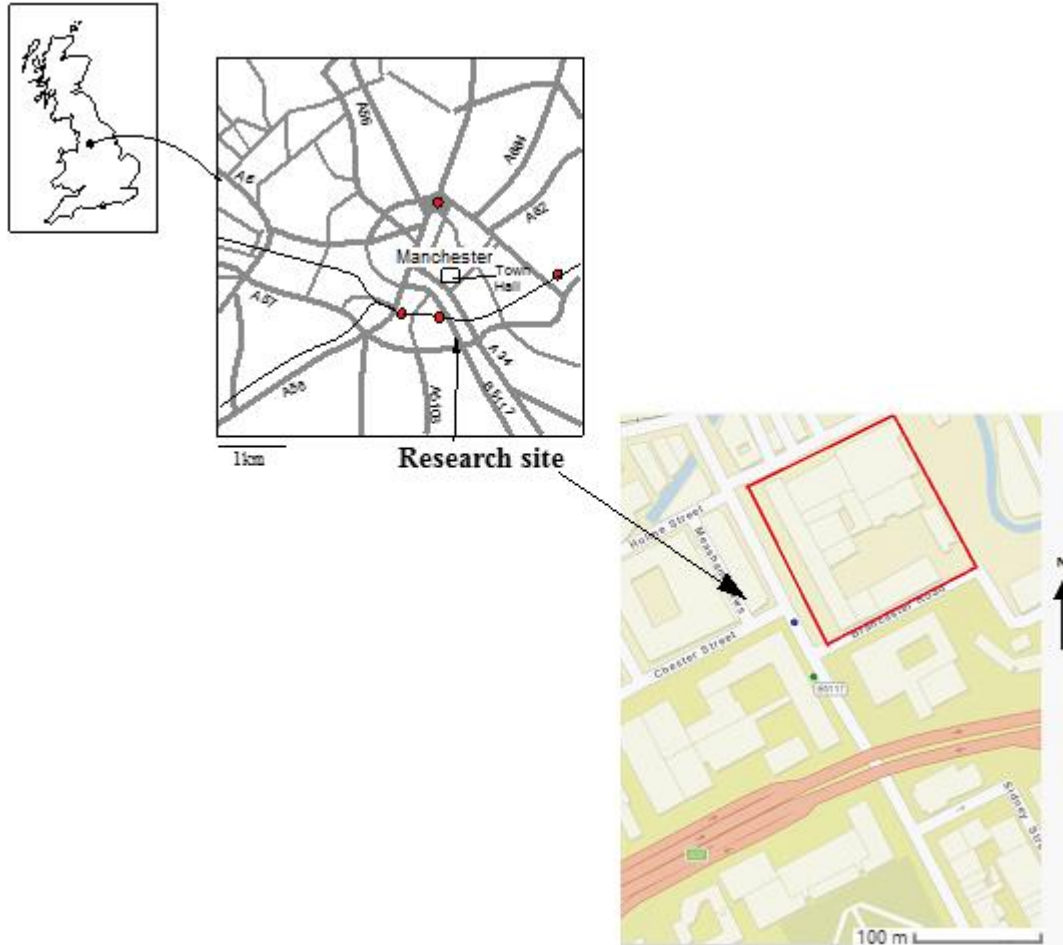
7

8   **Figure 1**: **Sediment sampling site on Oxford Road, Manchester, UK**

9   *2.3   Digestion of grain-specific fractions*

10   Three replicates of 0.25g ± 0.0015g for each sample was weighed into a polytetrafluoroethylene microwave vessel
11   and extracted with 1.5 ml of hydrofluoric acid, 5 ml of nitric acid and 2 ml of hydrogen peroxide, all reagent grade.
12   This was then subject to an initial digestion using a 1600W CEM Mars 5 Microwave, maximum temperature 200°C.
13   12ml of boric acid was then added to assist a further microwave-assisted digestion at a maximum temperature of
14   80°C. The aim of the secondary digestion is to precipitate excess fluoride and prevent secondary reactions (Dulski
15   1996). The samples were then cooled and diluted appropriately having been filtered using 0.45μm polyethersulfone
16   syringe filters. This method is based on the Environment Agency's EN14385 application note (BS EN 14385, 2004).
17   Use of this method has led to the accreditation of this lab by the United Kingdom Accreditation Service (UKAS)

18   *2.4   Geochemical analysis*

19   Digestion analytes of RD samples were analysed for 21 elements (Al, As, Ba, Ca, Cd, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, Pb,
20   Rh, S, Si, Sn, Ti, V and Zn) using a Varian Vista MPX ICP-OES with SeaSpray nebulizer. Initial system stability checks

1 were carried out with a 5 mg/l solution of manganese during the torch alignment whereby the upper values must
2 be in excess of 300,000 counts per second in both the axial and radial configurations, although sample analysis
3 exclusively used the axial position. A Four point linear calibration was achieved for each element, a calibration
4 coefficient of >0.95 for all elements was determined. The maximum concentrations of calibration points were
5 100mg/l for major elements, 20mg/l for minor and trace elements, and 1000mg/l for sulphur alone. Multiple
6 wavelength tests were taken to ensure high intensity with little interference was achieved. A total of five blank
7 samples as well as two extractions of MESS-3 certified reference material were also analysed to ensure quality
8 results.

9 ### 2.5 PM_{10} Concentrations

10 As indicated in figure 1, there is an air sampling station maintained by Manchester City Council near the demolition
11 site, the site is referred to as MAN1. The monitoring site quantifies airborne particulate matter less than 10µm in
12 size (PM_{10}) hourly and operates continuously. The site is approximately 40 meters south of the demolition site on
13 the opposite side of Oxford Road. The data were archived and available to the public via the Air Quality England
14 website. For the purpose of this study, archived data were reviewed to assess the changes in airborne particulate
15 matter in the direct vicinity of the demolition site. Average PM_{10} Concentrations were compared over the November
16 to May winter periods for winter 2010-11, 2011-12, 2012-13 (demolition) and 2013-14.

17 ## 3. Results and discussion

18 ### 3.1 Geochemical Analysis

19 The concentrations of the 21 elements analysed in the road dust fractions for both sampling campaigns are
20 summarized in Table 2. It can be observed that the general trend for the major constituents is Si>Ca> Al, Fe, Mg> K,
21 Na, Ti. These figures are in agreement with literature from other studies in urban areas (Chen et al. 2012; Cesari et
22 al. 2012 and references therein). It is generally accepted that the large concentrations of these elements are mainly
23 due to their crustal abundance (Alomary and Belhadj 2007; Gunawardana et al. 2012). In general, trace element
24 concentrations are slightly more subject to variance because the sources are usually of an anthropogenic nature,
25 henceforth are highly dependent on the local sources of inputs (Amato et al. 2009a). In general, the same trends can
26 be observed in all size fractions for some of the trace elements as reported by Potgieter-Vermaak et al. (2012), i.e.
27 Zn>Cu>Pb>Cr for the two smaller fractions and Zn>Cu> Cr> Pb for the largest fractions. All trace elements (indicated
28 with cursive font in Table 2) displayed the highest concentration in the finest fraction with a steady decrease in
29 concentration with increase in particle size (as also reported by Bian and Zhu 2009 and Fujiwara et al. (2011)). In
30 addition, it is noted that the absolute concentrations are lower for the May sampling campaign and the variance in
31 concentration change can be observed in Figures 2 -4. Cr, Pb and Ni in particular are of concern to those exposed,
32 because of their severe toxic association (Najafi et al. 2009). There is low confidence in the results for As, Cd, Rh and
33 Sn due to the high %RSDs observed (>20%), therefore they will not be discussed in any great detail in this article.
34 The high %RSDs are due to the concentrations being close to the detection limit for the instrument used.
35
36 **Table 2.** Mean extractable concentrations of the 21 elements, and degree of variance expressed as standard
37 deviation determined for the three finest grain-size fractions collected from the sample site Oxford Road,
38 Manchester collected in November 2012 and May 2013 (n=3: values are quoted to one decimal place; relative
39 standard deviation (%RSD) to one decimal place in parenthesis)

Element	Elemental concentration (mg/kg)					
	<38µm		63-38µm		63-125µm	
	Nov 2012 (%RSD)	May 2013 (%RSD)	Nov 2012 (%RSD)	May 2013 (%RSD)	Nov 2012 (%RSD)	May 2013 (%RSD)
Al	44643.3 (5.7)	53672.0 (4.8)	31967.0 (5.1)	50099.3 (3.3)	34624.3 (3.5)	35948.7 (4.6)
As	11.1 (17.4)	9.1 (28.6)	8.2 (44.5)	4.3 (129.4)	9.6 (60.8)	3.6* (98.0)
Ba	*1069.0 (7.6)*	*853.7 (5.2)*	*972.7 (9.9)*	*798.3 (3.3)*	*685.7 (2.2)*	*581.7 (4.7)*

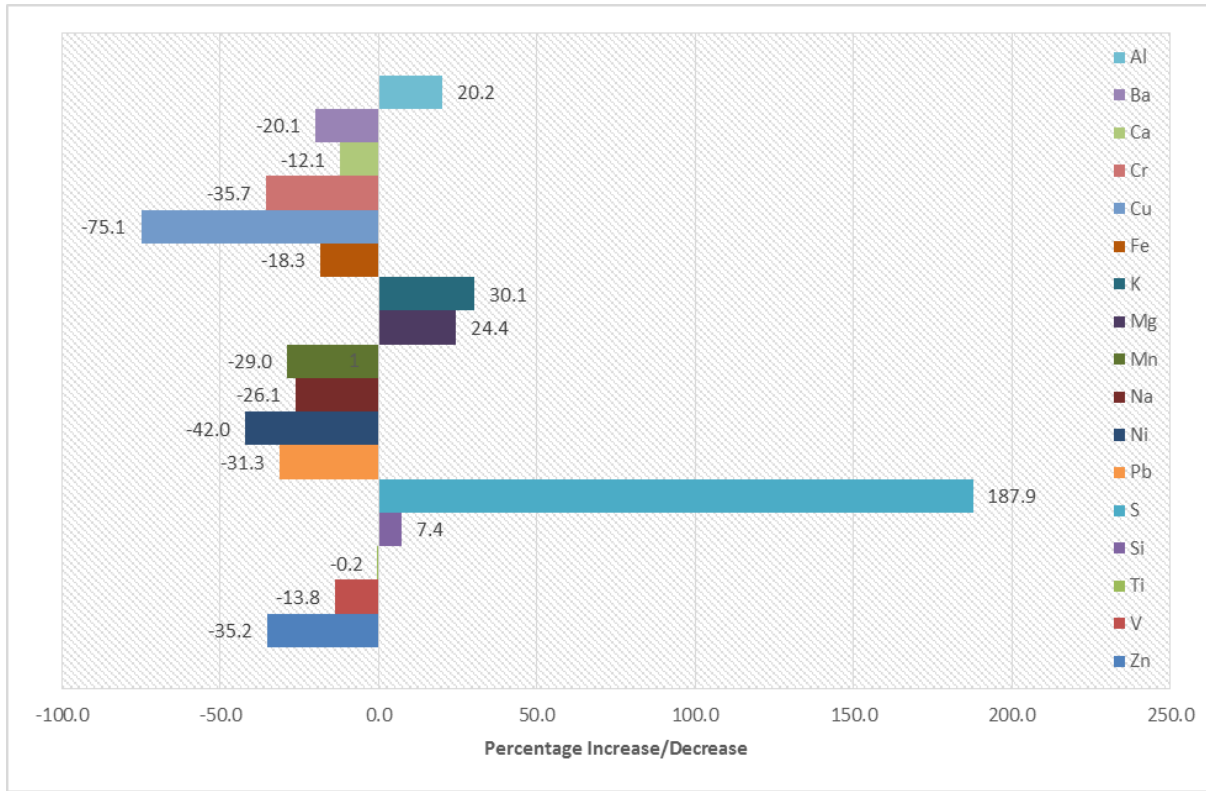|						
Ca	110397.0	97071.3 (5.1)	89352.3 (9.9)	89728.0 (2.2)	91271 (3.1)	78188.0 (4.1)
Cd	1.3* (79.7)	0.6* (55.4)	0.8* (138)	0.6* (71.3)	0.7* (19.7)	5.2 (16.5)
Cr	*324.7 (5.2)*	*208.7 (5.1)*	*311.0 (9.7)*	*208.7 (3.9)*	*274.7 (2.9)*	*181.3 (4.2)*
Cu	*1076.3 (8.6)*	*268.3 (9.3)*	*589.3 (10.6)*	*273.3 (3.0)*	*386.0 (3.1)*	*180.3 (7.8)*
Fe	38492.3 (5.6)	31456.3 (4.7)	29294.7 (6.7)	31374.3 (4.8)	28562.3 (3.5)	22501.7 (8.3)
K	15091.3 (9.6)	19636.3 (5.5)	14868.3	16987.7 (2.4)	14662.3 (4.1)	15310.7 (4.2)
Mg	21806.3 (8.5)	27129.7 (5.3)	16550.0 (9.3)	22256.0 (2.1)	14190.3 (3.0)	15621.0 (4.0)
Mn	*1250.7 (7.7)*	*887.7 (5.1)*	*1118.7 (9.7)*	*860.0 (2.9)*	*819.7 (2.5)*	*730.0 (4.4)*
Na	7933.3 (9.3)	5862.7 (6.6)	8194.3 (10.7)	6465.3 (2.4)	7245.3 (5.0)	7696.3 (4.1)
Ni	*122.0 (6.7)*	*70.7 (3.9)*	*103.0 (7.6)*	*69.9 (1.6)*	*63.6 (5.3)*	*79.9 (4.4)*
Pb	*384.0 (8.7)*	*264.0 (5.1)*	*334.0 (9.7)*	*247.0 (2.1)*	*207.0 (3.8)*	*151.7 (7.9)*
Rh	32.9 (29.4)	27.0* (84.3)	23.0* (40.5)	6.5* (11.2)	15.6* (44.5)	34.9* (44.7)
Si	299319.3	321438.0	304412.0	322272.0	417087.3	452857.7
Sn	*62.4 (18.9)*	*13.3 (21.6)*	*45.9 (11.4)*	*13.5 (83.6)*	*22.2 (29.4)*	*7.4 (47.0)*
S	3898.8 (5.4)	11224.8 (6.5)	3494.4 (8.9)	9150.3 (1.5)	2092.8 (4.7)	3349.7 (2.9)
Ti	5112.3 (7.9)	5101.7 (4.7)	5140.3 (9.0)	4481.0 (2.8)	200.3 (21.0)	3349.3 (4.3)
V	*115.7 (6.6)*	*99.7 (6.5)*	*104.9 (10.7)*	*95.4 (4.1)*	*77.4 (0.0)*	*76.0 (3.6)*
Zn	*1184.7 (8.5)*	*767.7 (6.6)*	*1017.0 (11.0)*	*624.7 (2.8)*	*614.0 (3.8)*	*384.7 (6.4)*

1 * Denotes replicates below detection limit, random numbers below detection limit have been inserted to obtain a
2 value for the average concentration

3
4 *3.2   Change in Road Dust Character over Sampling Period*

5 The key aim of this investigation is to assess the change in composition of inhalable particles due to building
6 demolition and assess the risk change on human health. This section discusses the observed changes in the elements
7 in terms of percentage change between the two sampling campaigns (November 2012, May 2013) that covers the
8 period of demolition. The percentage changes in each of the specified elements from the sampling campaign in
9 November 2012 to May 2013 are presented in figures 2-4. Each fraction is presented separately.

Figure legend (right side):
- Al
- Ba
- Ca
- Cr
- Cu
- Fe
- K
- Mg
- Mn
- Na
- Ni
- Pb
- S
- Si
- Ti
- V
- Zn

Data labels: 20.2, -20.1, -12.1, -35.7, -75.1, -18.3, 30.1, 24.4, -29.0, 1, -26.1, -42.0, -31.3, 187.9, 7.4, -0.2, -13.8, -35.2

X-axis: Percentage Increase/Decrease (-100.0, -50.0, 0.0, 50.0, 100.0, 150.0, 200.0, 250.0)

1
2

3   **Figure 1**. Percentage increase/decrease of each element in <38μm fraction relative to initial concentration from
4   November 2012 sample.

5
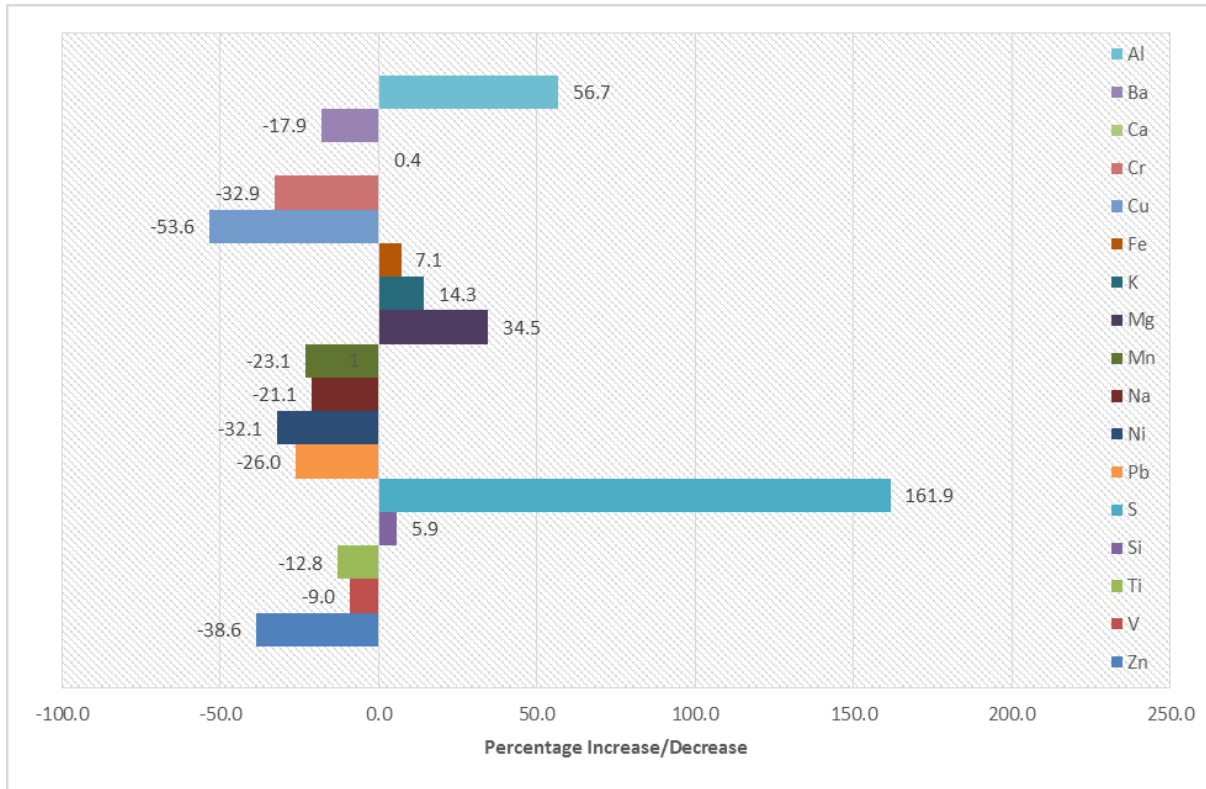
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2 **Figure 2.** Percentage increase/decrease of each element in 63-38μm fraction relative to initial concentration from
3 November 2012 sample.

4
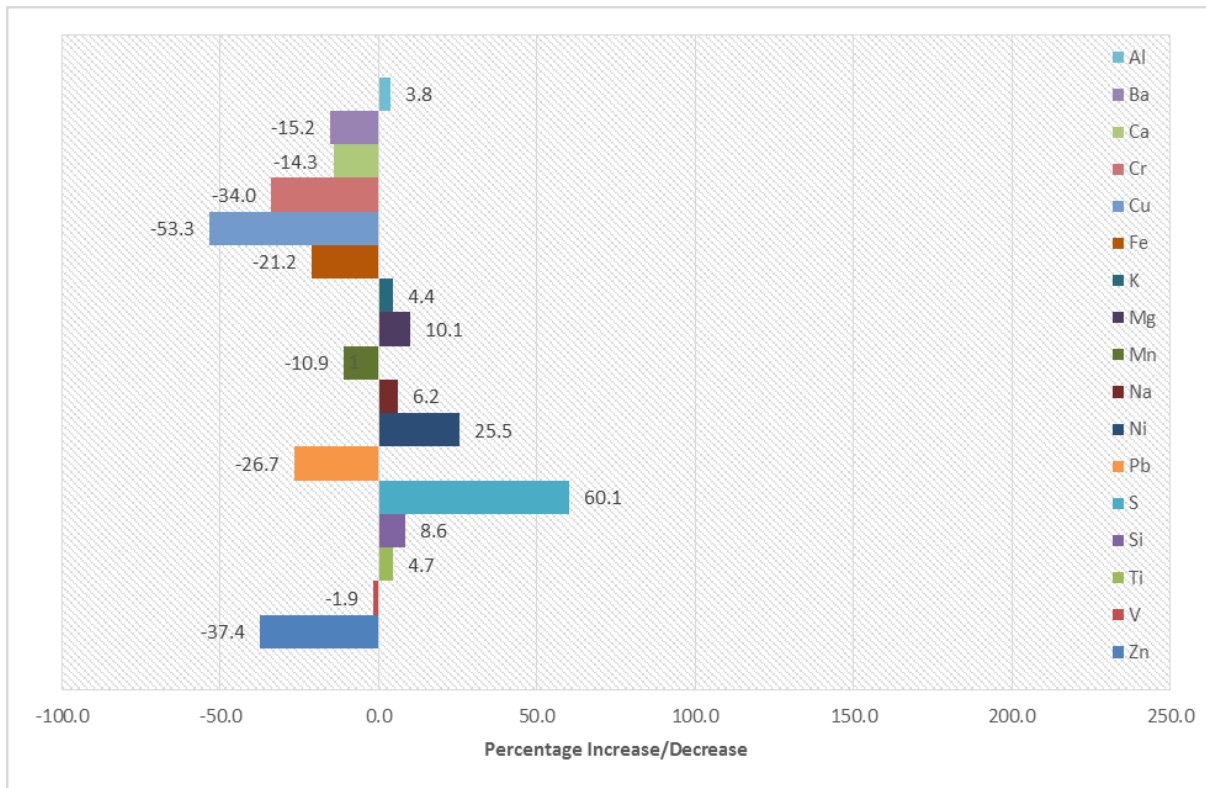
Figure 3. Percentage increase/decrease of each element in 125-63μm fraction relative to initial concentration from November 2012 sample.

3.2.1 Elemental Concentration Decreases

It is noted that around 60% of the elements investigated, namely Ba, Ca, Cr, Cu, Fe, Mn, Na, Ni, Pb, Ti, V and Zn, showed a decrease in their elemental concentrations between the two sampling periods. Interestingly the average decrease of these elements exhibited per fraction was fairly consistent at -26 ±2.2%. Across the three grain-size fractions the greatest decreases are seen for Cu. Scrutiny of percentage change on a grain-size basis finds that for the <38 μm fraction Cu > Ni > Cr>Zn > Pb, whilst decreases in element concentrations for the 63-38 μm and 125-63 μm fractions are found to be Cu > Zn > Cr > Pb. The behaviour of Ni in the two coarsest fractions is contradictory, whilst exhibiting similar decreases to that of Cr (32-33%) in the 63-38 μm fraction, the concentration Ni is seen to increase (25%) in 125-63 μm fraction.

A decrease in elemental concentrations of sediments within the literature is often attributed to increased rainfall (Jackson et al. 2007 and references therein). Increased rainfall during the months from December to February, might explain the lower concentrations of the 10 metals achieved in the May sampling campaign. In addition, it is possible that the significant increase in S observed from the building site, during the sampling period contributed to the diminishment of most other elemental species. The sulphurous nature of the Manchester RD coupled with the rain events had the potential to lower the pH of the RD and consequently reduce the inherent buffering capacity of the RD (Du et al. 2014), leading to metals being leached from the RD in the surface run-off into the urban water system, similar processes have been observed in soils by Xie (2004). This suggests that increase in the concentration of a contaminant, such as S, may have led to changes in the intended environmental receptor, with notable decreases in the concentrations of contaminants such as Cr, Cu, Ni, Pb, V and Zn available for inhalation, and corresponding enrichment of urban waters of these potentially harmful elements.

3.2.2 Elemental Concentration Increases

S, Si, Mg, K, Al increased in concentration, in all fractions, between the sampling periods. In contrast to the elemental decreases, the average increase for these elements did not stay constant (36.5% ± 19.5%). Moreover, the average increase for these elements was 54, 40 and 15% for the <38, 63–38, and 125-63 μm fractions, respectively. This clearly shows a significant difference in behaviour between the 3 fractions with reference to elemental concentration increase.

Application of the Mann-Whitney U-test find that the increases between Nov 2013 and May 2014 sampling campaign were significant (>95% confidence) for S, K and Al but not Si and Mg. This suggests an additional source of such elements has occurred during between the two sampling campaigns. We could speculate that the large increases in sulphur concentration (187.9% in the <38μm fraction, 161.9% in the 63-38μm fraction and 60.1% in the 125-63μm fraction) are likely to come from building materials, most likely gypsum. The increased aluminium concentrations (20.2% in the <38μm fraction, 56.7% in the 63-38μm fraction and 3.8% in the 125-63μm fraction) are also likely to be due to use of aluminium in building materials.
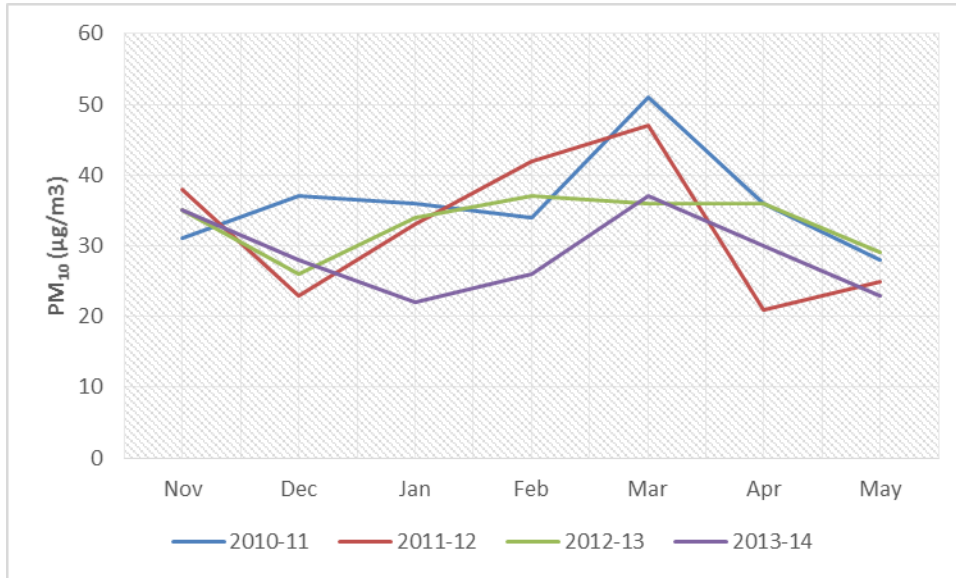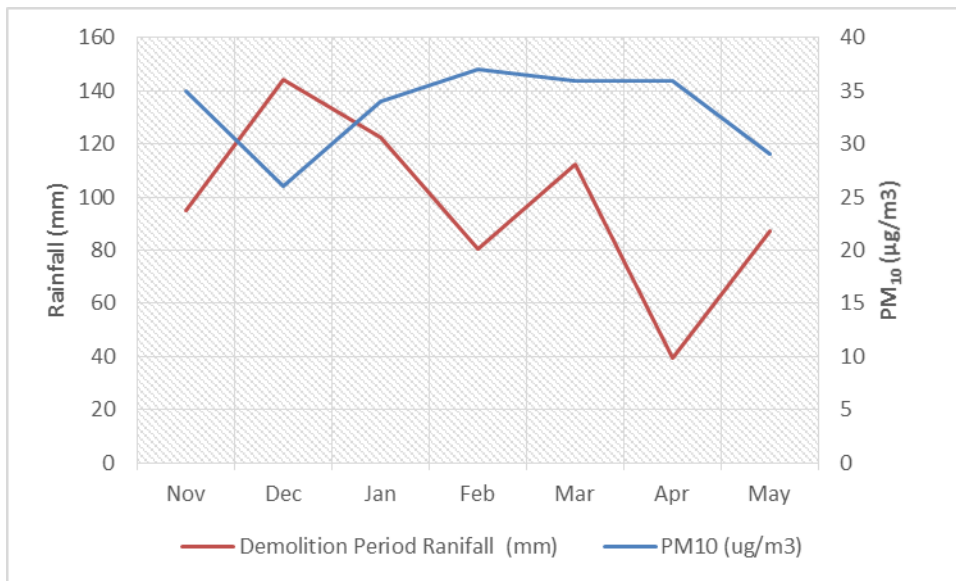
3.3 PM_{10} Concentrations

Results from the continuous monitoring site are presented in Figure 5. The monthly averages suggest that at the onset of the demolition (November) the PM mass concentrations were very similar to those observed in the previous 4 years. However, inspection of the data during March may lead to the conclusion that the spike in mass concentration observed during the previous and subsequent year during this time, is absent during the year of interest. Research in to sources of PM_{10} in an urban environment by Amato et al (2009b) indicated that PM_{10} concentrations increased by up to ten times, as a result of demolition work. The findings from this study appear to display different results. A possible reason for this could be the nature of the demolition, or perhaps a possibility could be the documented prevailing southerly wind in this area of Manchester (Lapworth and McGreggor 2008). It is likely that rainfall affected the concentration of PM_{10} during this period. Figure 6 represents monthly rainfall data

1 during the demolition period along with the demolition period PM_{10} data. Figure 6 indicates rainfall has an inverse
2 relationship with the concentration of PM_{10} during the months of the demolition as would be expected due to
3 washout. The negative correlation of -0.55 (p=0.05) displayed in figure 6 further supports this observation. This
4 influence of rainfall on PM is well documented (Shukla et al. 2008 and references therein), furthermore research
5 indicates that SO_2 and NO_2 is washed out of PM by precipitation (Davies 1976; Chang 1984). The washout could
6 potentially add to the acidification of urban runoff. All rainfall data is taken from Woodford Meteorological Station
7 16km from the sampling site (OS grid reference SJ895825).

8

9 **Figure 4.** Monthly average PM_{10} concentrations for winter and spring periods of 2010-11, 2011-12, 2012-13
10 (demolition took place in this period) and 2013-14

11

12 **Figure 5.** Monthly rainfall (mm) and concentration of PM_{10} during the demolition period

13 ***3.4   Risk Assessment***

14 Risk assessment has been used in many studies to assess the risk of contaminants in RD and road sediments (Du et
15 al, 2013; Ma and Singhirunnusorn 2012; Ferreira-Baptiste and De Miguel, 2005). This approach of risk assessment

1 requires the formulation of a conceptual site model to justify the need to assess a particular site. Table 3 presents
2 the heavily trafficked sample site both without and with the addition of the building demolition at the focus of this
3 paper.

4 **Table 3**: Conceptual site model showing ordinary exposure and sources, as well as exposure and sources due to the
5 building demolition, shown in italics

Source	Exposure Medium	Exposure Point	Exposure Route	Exposed Population
Road and Rail Traffic	Soils and Sediments	Oxford Road, Manchester	Ingestion, Inhalation and Dermal Contact	Commuters and Recreators
Building Demolition	*Soils and Sediments*	*Oxford Road, Manchester*	*Ingestion, Inhalation and Dermal Contact*	*Commuters and Recreators*

6

7 The next phase of risk assessment is then to calculate the exposure of an individual to any given pollutant.
8 Equation 1, 2 and 3 represent average daily dose (ADD) exposure via the Ingestion, Inhalation and dermal routes
9 respectively (USEPA 1997), ADD expressed in terms of mg/kg.day. The components and values used in these
10 equations are presented in table 4, as some of the components differ between an adult and a child, ADDs and
11 consequently HIs are different for adults and children. A child is assumed to be aged between 1 and 6 years old, in
12 accordance with USEPA (1989). It is therefore necessary to alter values for body weight, ingestion rate, inhalation
13 rate and skin surface area. The value for exposure duration is specific for this study as it represents the period of
14 demolition, six months.

15

16
$$ADD_{ing} = \frac{C \times R_{ing} \times CF \times EF \times ED}{BW \times AT}$$

17
$$ADD_{inh} = \frac{C \times R_{inh} \times EF \times ED}{PEF \times BW \times AT}$$

18
$$ADD_{derm} = \frac{C \times SA \times CF \times SL \times ABS \times EF \times ED}{BW \times AT}$$

19 **Table 4.** Parameters for risk assessment model

Component	*Definition (units)*	*Adult value used*	*Child value used*	*Reference*
C	Concentration of contaminant (mg/kg)	See table 2	See table 2	This study
R$_{ing}$	Ingestion rate (mg/day)	200	100	USEPA 1989
EF	Exposure frequency (days/year)	350	350	Du et al. 2014
ED	Exposure duration (years)	0.5	0.5	This study

BW	Body Weight (kg)	84	21	Potgieter-Vermaak et al. 2012
AT	Average Time (days)	365 x ED	365 x ED	Du et al. 2014
CF	Conversion factor (kg/mg)	1×10^{-6}	1×10^{-6}	
R_{inh}	Inhalation rate (m^3/day)	20	5	Du et al. 2014
PEF	Particle emission factor (m^3/kg)	1.32×10^9	1.32×10^9	USEPA 2000
SA	Surface area (cm^2)	5000	1800	Du et al. 2014
SL	Skin adherence factor (mg/cm^2)	1	1	Du et al. 2014
ABS	Dermal absorption factor	1×10^{-3}	1×10^{-3}	Du et al. 2014

1

2 A hazard quotient (HQ) can then be calculated by dividing the ADD by a reference dose (Rdf) specific to a chemical.
3 The reference dose acts as a maximum exposure below which the health concerns of the chemical are not a concern.
4 Therefore a HQ>1 is not considered a health concern. HQs from each of the three exposure routes can then be
5 summed to give a hazard index value (HI), again, a value below 1 is not considered a concern (USEPA 2001).

6 Tables 5 and 6 present the HI results of the risk assessment model for adults and children respectively. It is observed
7 that the HI values do not exceed 1 for any element, with the exception of chromium exposure in children. Full tables
8 showing Rdf, ADD and HQ values for each of the exposure routes can be found in the supplementary section of this
9 paper, listed as tables S1 and S2 respectively. All Rdf values are taken from literature (Liu et al. 2015; Cao et al. 2015;
10 Buranatrevedh 2013; HEAST 2011)

11 **Table 5.** HI values for adults for each element from the three exposure routes, using USEPA PEF value

	Al	*Ba*	*Cr*	*Cu*	*Pb*	*Ni*	*V*	*Zn*
HI	**0.147**	**0.005**	**0.524**	**0.008**	**0.039**	**0.004**	**0.014**	**0.004**

12

13 **Table 6.** HI values for children for each element from the three exposure routes, using USEPA PEF value

	Al	*Ba*	*Cr*	*Cu*	*Pb*	*Ni*	*V*	*Zn*
HI	**0.575**	**0.039**	**1.076**	**0.062**	**0.099**	**0.032**	**0.102**	**0.024**

14

15 One assumption of the ADD calculations used here is the PEF parameter from the inhalation calculation. Despite
16 being commonly used in literature to assess the risks associated with RD, the PEF value, expressed as m^3/kg, was
17 derived for the purpose of representing airborne quantities of soil contaminants, and relies on the site having a 50%
18 vegetation coverage (USEPA 2000). The applicability of this PEF value to samples of this nature is therefore
19 questionable, as the physical and morphological characteristics of RD differ greatly to soil, also the vegetation cover
20 at the site used in this study is 0%. As previously noted in the introduction section of this paper, RD has been shown

1   to account for 62% of TSP (Harrison et al. 1997) and 74% of TSP (Hien et al., 1999). Having monitored TSP
2   concentrations near the demolition site for over a year as part of another study, the average concentration of TSP is
3   92.9µg/m³, we can therefore estimate that a lower bound estimate of the quantity of TSP derived from road dust is
4   62%. Replacing the PEF value in the ADD_{inh} equation with 62% of 92.9µg/m³ could therefore serve as another
5   estimate of ADD_{inh}, it also stands to make the equation used more specific for this study site. To maintain the
6   mg/kg.day units for ADD_{inh}, the ADD_{inh} equation is altered as displayed below. Here, the units for TSP are in kg/m³,
7   all other parameters are the same for the USEPA ADD_{inh} equation.

8
$$ADD_{inh} = \frac{C \times R_{inh} \times EF \times ED \times ([TSP] \times 0.62)}{BW \times AT}$$

9

10   A new risk assessment for adults and children table can then be derived using this equation for ADD_{inh} whilst keeping
11   the other two ADD equations, the HI results are presented in tables 7 and table 8 respectively. Again, full tables
12   showing ADD and HQ values for each of the exposure routes can be found in the supplementary section of this paper,
13   listed as tables S3 and S4 respectively.

14   **Table 7**. HI values for adults for each element from the three exposure routes, using the amended PEF value for the
15   ADD_{inh} calculation

	Al	Ba	Cr	Cu	Pb	Ni	V	Zn
HI	0.711	0.005	0.528	0.009	0.039	0.004	0.015	0.004

16

17   **Table 8.** HI values for children for each element from the three exposure routes, using the amended PEF value for
18   the ADD_{inh} calculation

	Al	Ba	Cr	Cu	Pb	Ni	V	Zn
HI	1.132	0.039	1.079	0.062	0.099	0.033	0.103	0.024

19

20   As a result of altering the PEF value, it can now be observed that the HI value for each element has increased, most
21   increases are very small and not sufficient to alter the three significant figures quoted. The most noteworthy change
22   is the HI value for aluminium exposure in children, which has increased from 0.575 to 1.13 and is therefore now
23   considered a health concern, along with chromium. The dramatic increase observed in the HI value for aluminium is
24   due to the high HQ value observed for the inhalation route. All HQ values increase by the same factor due to the
25   modified ADD_{inh} equation, however the proximity to 1 observed for the ADD_{inh} in the original equation causes a
26   greater effect on the HI when the equation is modified. The results of this study are comparable with the HI values
27   calculated by Du et al. (2013) with the exception of Al and Cr, which appear to give significantly larger HI values in
28   this study.

29

30   ### 4.   *Conclusion*

31   The design of this experimentation was based on the assumption that the concentrations of metals would increase
32   after the demolition of a building. However, in general this was not observed for all elements and 12 of the 17
33   elements of interest decreased by an average of 26% across all size ranges (Ba, Ca, Cr, Cu, Fe, Mn, Na, Ni, Pb, Ti, V
34   and Zn). A substantial increase in the finer two fractions was however observed (on average 54 and 40%) for the
35   remaining elements. As alluded to in the results and discussion section, it is likely that the large increase in
36   concentration of sulphur had an influence on the diminished concentrations of other metals that were observed.
37   Perhaps the main concern that can be raised from these results is the effect of the heavy enrichment in sulphur on

mobility of metals. While the increase in sulphur causes a reduction in contaminants available for resuspension, it increases the concentrations of contaminants in surface water. Moreover, it will add to the overall burden of urban land contamination due to metal enrichment of water run-offs and soil. Perhaps further investigation on contamination of surface water due to building demolition is required to assess the potential health effects associated. It is observed that rainfall and PM10 were inversely proportional during the demolition. This washout effect was also apparent in the RD, however to prove this conclusively, it may be necessary to compare rainfall, PM concentration and elemental concentrations over a larger period of time. Thus, one would be able to elucidate whether the elemental washout process occurs independently of the building demolition, or to what extent the building demolition affected the washout. The risk assessment portion of this analysis indicates that the HI for chromium and aluminium is a concern for children exposed at this site, when using the modified PEF value suggested.

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