

PAPER

View Article Online
View Journal | View Issue



Cite this: *Environ. Sci.: Atmos.*, 2022, 2, 1076

The oxidative potential of particulate matter (PM) in different regions around the world and its relation to air pollution sources†

Vahid Jalali Farahani,^a Abdulmalik Altuwayjiri,^{af} Milad Pirhadi,^b Vishal Verma,^c Ario Alberto Ruprecht,^d Evangelia Diapouli,^e Konstantinos Eleftheriadis^e and Constantinos Sioutas  ^{*a}

In this study, we investigated the impact of urban emission sources on the chemical composition of ambient particulate matter (PM) as well as the associated oxidative potential. We collected six sets of PM samples in five urban location sites around the world over long time periods varying from weeks to months, intentionally selected for their PM to be dominated by unique emission sources: (1) PM_{2.5} produced mainly by traffic emissions in central Los Angeles, United States (US); (2) PM_{2.5} dominated by biomass burning in Milan, Italy; (3) PM_{2.5} formed by secondary photochemical reactions thus dominated by secondary aerosols in Athens, Greece; (4) PM₁₀ emitted by refinery and dust resuspension in Riyadh, Saudi Arabia (SA); (5) PM₁₀ generated by dust storms in Riyadh, SA, and (6) PM_{2.5} produced mainly by industrial and traffic emissions in Beirut, Lebanon. The PM samples were chemically analyzed and their oxidative potential were quantified by employing the dithiothreitol (DTT) assay. Our results revealed that the Milan samples were rich in water soluble organic carbon (WSOC) and PAHs, even exceeding the levels measured on Los Angeles (LA) freeways. The PM in Athens was characterized by high concentrations of inorganic ions, specifically sulfate which was the highest of all PM samples. The ambient PM in LA was impacted by the traffic-emitted primary organic and elemental carbon. Furthermore, the contribution of metals and elements per mass of PM in Riyadh and Beirut samples were more pronounced relative to other sampling areas. The highest intrinsic PM redox activity was observed for PM with the highest WSOC fraction, including Milan (biomass burning) and Athens (secondary organic aerosols, SOA). PM in areas characterized by high metal emissions including dust events, refinery and industry, such as Riyadh and Beirut, had the lowest oxidative potential as assessed by the DTT assay. The results of this study illustrate the impact of major emission sources in urban areas on the redox activity and oxidative potential of ambient PM, providing useful information for developing efficient air pollution control and mitigation policies in polluted areas around the globe.

Received 11th April 2022

Accepted 2nd July 2022

DOI: 10.1039/d2ea00043a

rsc.li/esatmospheres

Environmental significance

Particulate matter (PM) is one of the main atmospheric drivers for adverse health effects on living organisms. However, the potential toxicity of ambient PM is not statistic variable, but rather a dynamic one and a function of its chemical composition and the emission source. In this study, we evaluated the chemical composition and the corresponding oxidative potential of ambient PM in various metropolitan areas, providing useful information for the impact of the urban emission source on the redox activity of ambient PM. This information can be utilized to develop efficient air pollution control policies in urban areas around the globe.

^aUniversity of Southern California, Department of Civil and Environmental Engineering, 3620 S. Vermont Ave, KAP210, Los Angeles, California 90089, USA. E-mail: sioutas@usc.edu; Fax: +1-213-744-1426; Tel: +1-213-740-6134

^bCalifornia Air Resources Board, Sacramento, California, USA

^cUniversity of Illinois at Urbana Champaign, Department of Civil and Environmental Engineering, Urbana, Illinois, USA

^dInternational Society of Doctors for Environment (ISDE), Italy

^eEnvironmental Radioactivity Laboratory, N.C.S.R. Demokritos, 15341 Attiki, Greece
^fMajmaah University, Department of Civil and Environmental Engineering, Majmaah, Riyadh, Saudi Arabia

† Electronic supplementary information (ESI) available. See <https://doi.org/10.1039/d2ea00043a>



Six batches of PM samples with varied particle size fractions (*i.e.* PM_{2.5} or PM₁₀) and sampling periods were the focus of this synthesis study. The PM samples were collected in cities of Los Angeles, Milan, Athens, Riyadh, and Beirut, each corresponding to unique set of PM emission sources. Table 1 summarizes the relevant information for each sampling batch. The samples in Riyadh were collected during two periods, one corresponding to dust storm events to measure the impact of dust particles on the oxidative activities and the other corresponding to a non-dust period to investigate the contribution of refinery emissions and dust resuspension to the PM oxidative potential. The samples in Athens were collected during the summertime when the rate of photochemical oxidation is at peak, enhancing the

Environ. Sci.: Atmos., 2022, **2**, 1076–1086 | **1077**

content of secondary organic aerosols (SOA) as the main PM emission source in that region. Furthermore, the high temperature in the summer minimizes the contribution of primary volatile and semi-volatile compounds to PM mass. Biomass burning is another major source generating PM redox activity in the metropolitan areas in the colder time period of the year. We investigated the impact of this emission source on the PM oxidative potential by incorporating the samples collected in Milan during the winter. According to our previous investigation, the overall oxidative potential of Milan's ambient particles during cold periods is largely induced by the biomass burning activities in this urban region.⁴¹ The ambient PM in cities of Los Angeles and Beirut are both heavily impacted by the traffic emissions. However, they are dissimilar in their PM chemical composition. The Beirut PM redox activity is dominated by the transition metals, compared to the LA area, in which organic compounds are the driving components in generating redox activity.²⁸ Furthermore, particles in Beirut are also influenced by industrial emissions to a much higher degree than those in Los Angeles.

2.2. Chemical analysis

The PM samples were chemically analyzed for their content of elemental carbon (EC), organic carbon (OC), water-soluble inorganic ions, polycyclic aromatic hydrocarbons (PAHs) as well as metals and trace elements by the Wisconsin State Lab of Hygiene (WSLH). The EC/OC content in the PM samples were measured *via* the Thermal Evolution/Optical Transmittance (TOT) analytic method using a model-4-semi-continuous OC/EC field analyzer (Sunset Laboratory Inc, USA).⁴² A small fraction of the filter was punched and kept in an oven, which was set at a temperature of 820 °C to capture the oxidized OC fraction. The oven temperature was decreased and then raised to the temperature of 860 °C through the mixture of helium and oxygen gases to measure the oxidized EC fraction. The WSOC fraction of PM samples was quantified through the extraction of PM in ultrapure water and the subsequent filtration (0.22 µm pore size) of the samples using a Sievers 900 total organic carbon (TOC) analyzer (GE Analytical Instruments, Boulder, CO, USA).⁴³ The analysis of PM-bound metals and trace elements was performed by means of inductively coupled plasma-mass spectroscopy (ICPMS).^{44,45} A punch of the filter was digested in a mixture of nitric acid, hydrochloric acid, and hydrofluoric acid (0.6 mL 16 N HNO₃, 0.2 mL 12 N HCl, 0.1 mL 28 N HF) using an automated, temperature- and pressure-regulated, trace analysis microwave system (Milestone Ethos+). The PAH concentrations were measured through chromatography/mass spectrometry (GC/MS) analysis as described in Sánchez *et al.*⁴⁶ The remaining part of the filters were extracted in a mixture of ultrapure water and ethanol by means of ion chromatography using a Dionex Model DX-500 Ion Chromatograph to quantify the PM inorganic ions content including ammonium (NH₄⁺), nitrate (NO₃⁻), and sulfate (SO₄²⁻).

2.3. PM oxidative potential

Oxidative potential of the PM samples was determined by employing an *in vitro* DTT assay.^{47,48} The DTT is one of the most

commonly used acellular assays to quantify PM OP.^{23,49} The OP measured by the DTT assay has shown a stronger association with wide range of adverse health effects compared to PM_{2.5} mass concentration^{50–53} and therefore is a suitable metric to represent the health effects associated with PM exposure. Furthermore, past studies have shown that the DTT-based OP is sensitive to chemical composition,^{16,32,47,54} changes seasonally^{55–57} and is extremely responsive to episodic events such as wildfires,⁵⁸ haze⁵⁹ and fireworks.⁶⁰

The quantification of PM oxidative potential using this assay involves the incubating DTT with PM aqueous extracts, and the DTT depletion rate is measured, which is proportional to the generation rate of PM-driven ROS. The PM samples were stored in a freezer at a temperature of –20 °C and extracted into an aqueous media using high purity Milli-Q water. The extracts were incubated in the mixture of potassium phosphate (KPO₄) buffer and DTT, and then the linear rate of DTT consumption was quantified. Further details of the DTT methodology are available elsewhere.⁵⁶ The intrinsic and extrinsic DTT values were estimated by normalizing DTT depletion rate to PM mass and sampled air volume, respectively. The intrinsic DTT values are in units of nmoles per min per mg of PM and reflect the oxidative potential of PM per unit mass, while the extrinsic DTT values are in units of nmol per min per m³ of air and indicate the oxidative potential per unit volume of air of the aerosol component. The consistency of the measured DTT was evaluated by positive control as well as multiple field blanks, which were included in every sample queue. The standard chemical employed for positive control was 9,10-phenanthraquinone (PQ) based on the reported sensitivities.^{23,49} The concentration of the PQ was 0.2 mM in the reaction vial of DTT while the average rate of positive control was 1.867 ± 0.094 µM min⁻¹ or 0.037 ± 0.039 pmol (min µg)⁻¹.⁶¹

3. Results and discussion

3.1. Chemical composition of PM

The PM chemical constituents were investigated by organic carbon (OC) and the associated water-soluble organic carbon (WSOC), elemental carbon (EC), metal and trace elements as well as inorganic ions including chloride (Cl⁻), nitrate (NO₃⁻), sulfate (SO₄²⁻), sodium (Na⁺), ammonium (NH₄⁺) and potassium (K⁺). The total metallic content was calculated as the sum of the crustal and trace elements. The crustal portion of the metal elements was quantified using the following equation which is based on the oxidized form of metal elements:^{62,63}

$$C = 1.89Al + 1.21K + 1.43Fe + 1.40Ca + 1.66Mg + 1.70Ti + 2.14Si \quad (1)$$

Silicon (Si) is not determined in the ICP-MS analysis; thus, the concentration of this specie was calculated as 3.41 times of aluminum concentration (Al).⁶² The average mass fraction of individual species for PM sample sets at each location is shown in Table 2.

The PM_{2.5} sample sets in Milan were dominated by water-soluble inorganic ions (380.71 µg mg⁻¹ PM) and OC (217.76



Table 2 The mass fraction of PM chemical components in the studied cities^a

	Beirut	Athens	Los Angeles	Riyadh (dust)	Riyadh (non-dust)	Milan
PM constituents ($\mu\text{g mg}^{-1}$ PM)						
EC	10.04	13.36	23.05	6.71	23.00	28.05
OC	90.27	241.64	241.12	34.62	76.55	217.76
WSOC	51.98	147.40	68.62	12.86	44.08	118.79
Cl ⁻	0.13	1.09	—	4.56	6.23	9.74
NO ₃ ⁻	0.94	3.93	179.15	20.73	40.45	224.75
SO ₄ ²⁻	93.41	264.17	56.41	37.37	77.97	40.29
Na	3.26	15.18	—	4.30	7.07	3.03
NH ₄ ⁺	31.89	69.57	46.83	2.67	8.33	91.05
K ⁺	2.17	5.79	—	2.07	2.87	11.85
Metals (ng mg ⁻¹ PM)						
Ca	58 088.44	44 540.28	19 023.33	134 400.06	160 764.66	10 754.79
Al	13 476.45	18 532.70	7729.16	58 536.25	40 047.85	5424.49
Fe	8540.81	18 755.25	7333.21	39 446.27	28 165.42	5973.15
Mg	17 613.01	8073.74	9885.59	28 090.01	18 773.74	4964.33
Zn	4211.02	1348.21	453.39	221.65	2157.01	618.18
Ba	537.62	451.12	478.53	430.75	538.88	214.18
Cu	801.49	281.73	262.63	84.59	218.36	241.26
Ti	849.11	989.80	481.08	3848.63	2806.80	119.48
Mn	1032.63	387.01	118.95	758.83	562.91	158.37
Pb	210.67	259.08	55.92	46.60	185.52	293.95
Ni	586.03	145.63	47.45	95.74	70.91	37.14
Sn	56.92	134.82	71.62	11.14	33.22	147.62
Cr	199.16	203.04	86.14	118.88	92.63	111.43
V	29.06	132.10	17.78	112.59	97.13	6.60
Li	31.34	8.46	15.91	33.14	20.91	4.17
Cd	5.67	6.57	1.22	1.01	3.11	4.84
Pd	0.65	0.56	2.34	1.09	0.62	4.52

^a Redox-active metals are highlighted in the table.

$\mu\text{g mg}^{-1}$ PM), followed by trace amounts of metals (119.57 $\mu\text{g mg}^{-1}$ PM). High levels of water-soluble organic carbon (WSOC) (118.79 $\mu\text{g mg}^{-1}$ PM), a tracer of biomass burning during the cold periods in the absence of photochemistry,^{64–66} were observed in Milan as a result of significant biomass burning in that city during the sampling period.⁴¹ Nitrate (NO₃⁻) and ammonium (NH₄⁺) constituted the highest fraction of PM among inorganic ions (224.75 and 91.05 $\mu\text{g mg}^{-1}$ PM, respectively), which further corroborates the impact of biomass burning on PM levels. This is also consistent with Daellenbach *et al.*³ findings in Europe, attributing high SOA levels in winter to the oxidation of the anthropogenic precursors, mainly from biomass burning activities.

Similar to Milan, the ambient fine particles in Athens displayed high levels of OC (241.64 $\mu\text{g mg}^{-1}$ PM) and water-soluble inorganic ions (359.74 $\mu\text{g mg}^{-1}$ PM). Markers of secondary organic aerosols (*i.e.*, SO₄²⁻ and NH₄⁺) were among the highest of all PM batches and dominated Athens's water-soluble inorganic ions, underscoring the impact of SOA on ambient PM loadings in this region. Furthermore, the WSOC per mass value was 147.40 $\mu\text{g mg}^{-1}$ PM, a significant increase from that of measured in Beirut (51.98 $\mu\text{g mg}^{-1}$ PM) and Riyadh during dust storms (12.86 $\mu\text{g mg}^{-1}$ PM) and non-dust periods (44.08 $\mu\text{g mg}^{-1}$ PM). The samples in Riyadh exhibited significant loadings of PM-bound metals during both dust (299.63 $\mu\text{g mg}^{-1}$ PM) and non-dust (300.78 $\mu\text{g mg}^{-1}$ PM) periods. The chemical

composition of ambient PM_{2.5} in Los Angeles was primarily composed of carbonaceous compounds. The quantified EC (23.05 $\mu\text{g mg}^{-1}$ PM) and OC (241.12 $\mu\text{g mg}^{-1}$ PM) were among the highest of the studied sites in this heavily trafficked region, while WSOC constituted relatively small fraction (68.6 $\mu\text{g mg}^{-1}$ PM) of the total organic carbon. Nitrate was the dominant specie (179.15 $\mu\text{g mg}^{-1}$ PM) among the water-soluble inorganic ions, followed by sulfate (56.41 $\mu\text{g mg}^{-1}$ PM) and ammonium (46.83 $\mu\text{g mg}^{-1}$ PM). Moderate loadings of total metals (160.14 $\mu\text{g mg}^{-1}$ PM) were also observed in the LA basin which have been mostly associated with the traffic sources including resuspended dust and the vehicle abrasion.^{35,67}

Table 3 compares the PAH levels observed in Beirut, Athens, LA, Riyadh and Milan. The PAHs per PM mass concentration in Milan were substantially higher than other location sites across all PAH species. The PAHs are carcinogenic organic compounds which are typically released in particle phase during incomplete combustion from gasoline and diesel-fueled vehicles as well as biomass burning.^{68–72} The PAH content of the Milan samples, which are mainly driven by biomass burning as noted earlier, were significantly greater compared to the corresponding ambient levels in LA basin where PM is primarily released from vehicle emissions. In fact, the observed values in Milan even exceeded our previously measured PAH values inside two major Los Angeles's freeways,⁷³ as shown in Fig. 1. The cumulative PAH content in Milan were 1.35 ng μg^{-1} PM which is greater



Table 3 The mass fraction of PAH components in the studied cities^a

PAH species (ng μg^{-1} PM)	Beirut	Athens	LA	Riyadh (dust)	Riyadh (non-dust)	Milan
Phenanthrene	<LOD	0.0076	0.0030	0.0002	0.0042	0.0124
Fluoranthene	0.0010	<LOD	0.0026	0.0004	0.0075	0.0462
Pyrene	0.0008	<LOD	0.0031	0.0002	0.0070	0.0433
Benzo(<i>ghi</i>)fluoranthene	<LOD	<LOD	0.0005	0.0001	0.0048	0.0945
Benz(<i>a</i>)anthracene	<LOD	<LOD	0.0042	<LOD	<LOD	0.0623
Chrysene	<LOD	<LOD	0.0002	0.0001	0.0047	0.2253
Benzo(<i>b</i>)fluoranthene	<LOD	<LOD	0.0109	0.0007	0.0249	0.2402
Benzo(<i>k</i>)fluoranthene	<LOD	<LOD	0.0016	0.0001	0.0040	0.2214
Benzo(<i>f</i>)fluoranthene	<LOD	<LOD	<LOD	<LOD	<LOD	0.0080
Benzo(<i>e</i>)pyrene	<LOD	<LOD	0.0090	0.0006	0.0175	0.1665
Benzo(<i>a</i>)pyrene	<LOD	<LOD	<LOD	<LOD	<LOD	0.0054
Indeno(1,2,3- <i>cd</i>)pyrene	<LOD	<LOD	<LOD	<LOD	0.0144	0.0665
Benzo(<i>g,h,i</i>)perylene	0.0003	<LOD	0.0006	0.0002	0.0087	0.0743
Coronene	<LOD	<LOD	<LOD	<LOD	0.0072	0.0217
Dibenzo(<i>a,e</i>)pyrene	<LOD	<LOD	<LOD	<LOD	<LOD	0.0016

^a The values below limits of detection (LOD) are shown as <LOD.

than that at I-110 (0.16 ng μg^{-1} PM) and I-710 (0.15 ng μg^{-1} PM) freeways by approximately one order of magnitude. These observations underscore the importance of biomass burning as one of the main driving factors in formation of PAHs. The low PAH levels in Athens during summer are probably the result of higher photo-degradation rate of these species with oxidizing gases (ozone, nitrogen oxides, hydrogen peroxide, *etc.*),^{74,75} as well as increased partitioning to the gas phase,⁷⁶ due to the higher ambient temperatures in that time of year. Similarly, Riyadh and Beirut are both Middle Eastern cities associated with very high temperature events during spring and summer, resulting in minor PAH levels for the reasons noted earlier. Fewer number of PAH species were detected in the coastal city of Beirut due to the frequent daytime on-shore winds at this site, which increases the atmospheric dilution and coupled with

the high temperature, it can enhance the volatilization, photo-degradation and dispersion of PAHs.²⁸

3.2. Comparison of PM oxidative potential among various locales

Table 4 shows the oxidative potential measured by the DTT assay normalized by PM mass (intrinsic DTT) as well as extrinsic DTT values. The data shown in Table 4 illustrate a wide range of oxidative responses measured by the DTT assay, underscoring the impact of location-specific emission sources and the resulting PM chemical composition on the PM oxidative stress. The lowest intrinsic level was measured in Beirut, where intrinsic DTT consumption rate was 8.22 ± 1.27 nmoles (min mg^{-1}). The PM sample sets collected in Athens and Milan

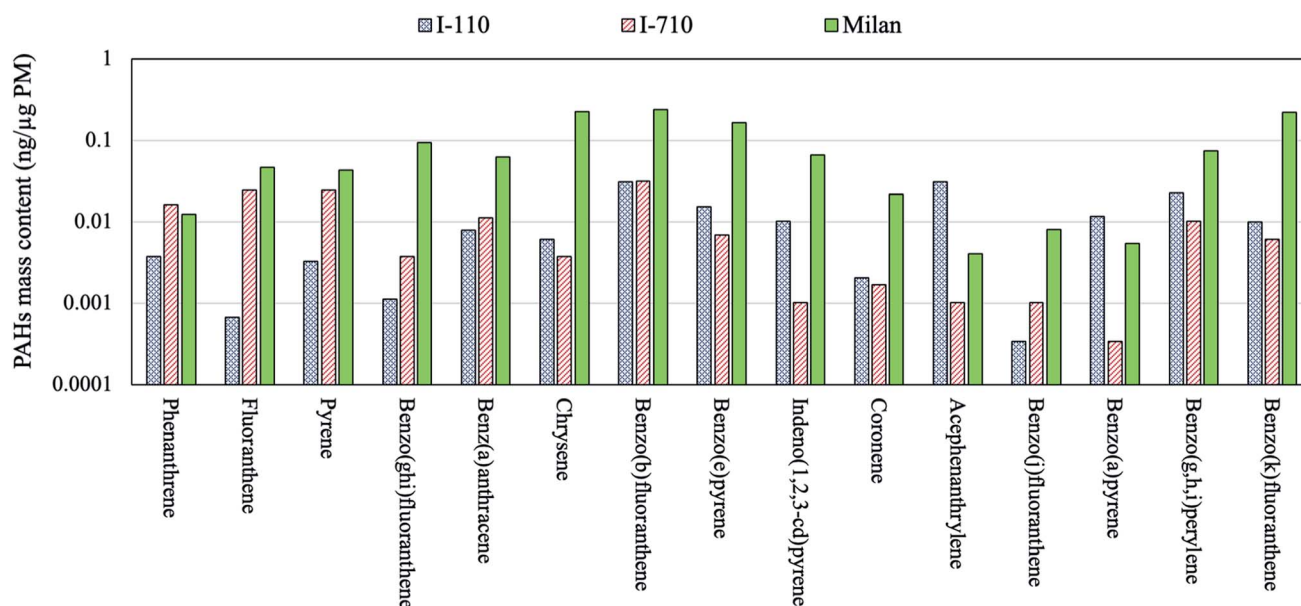


Fig. 1 The comparison of PAH levels in Milan and Los Angeles I-110 and I-710 freeways.



Table 4 The intrinsic and extrinsic DTT activity for the collected PM batches across the globe

Location	PM size	Intrinsic DTT (nmoles (min mg) ⁻¹)	Extrinsic DTT (nmoles (min m ³) ⁻¹)
Milan	PM _{2.5}	65.29 ± 5.17	3.38 ± 0.26
Athens	PM _{2.5}	49.20 ± 1.66	5.62 ± 0.19
Beirut	PM _{2.5}	8.22 ± 1.27	3.51 ± 0.54
Los Angeles	PM _{2.5}	28.10 ± 5.23	0.35 ± 0.04
Riyadh – dust period	PM ₁₀	9.32 ± 0.80	1.85 ± 0.15
Riyadh – non-dust period	PM ₁₀	12.53 ± 1.43	1.05 ± 0.12

exhibited 4–7 times higher DTT consumption rate, which in part can be attributed to the greater WSOC content of PM, originated from biomass burning and secondary aerosols, respectively in these cities. In agreement with this observation, Daellenbach *et al.*,³ who have investigated the sources of PM and oxidative potential across Europe, reported significant intrinsic oxidative potential from residential biomass burning activities. We will discuss the impact of different chemical constituents on the PM oxidative potential in detail in the following section. The intrinsic DTT values in Riyadh were somewhat higher for the PM collected during the non-dust period (12.53 ± 1.43 nmoles (min mg)⁻¹) relative to those collected in a dust event period (9.32 ± 0.80 nmoles (min mg)⁻¹). The local traffic emission sources in Los Angeles resulted in DTT activity of 28.10 ± 5.23 nmoles (min mg)⁻¹. The extrinsic DTT consumption rates spanned from 0.35 ± 0.04 nmoles (min m³)⁻¹ in LA to 5.62 ± 0.19 nmoles (min m³)⁻¹ in Athens. The different trends in volume-normalized DTT values compared to the mass-normalized approach can be attributed to the higher PM mass concentration in the respective samples, resulting into greater exposure to redox-active aerosols despite the lower capability of PM chemical constituents in inducing oxidative activity.

3.3. The impact of emission sources on PM oxidative potential

We performed a linear regression analysis between measured PM chemical constituents and the DTT activities at different sites to identify the emission sources linked to PM oxidative potential, the results of which are summarized in Table 5. In order to perform the regression analysis, we employed the average values for the PM chemical constituents (shown in Tables 2 and 3) and intrinsic DTT rates (Table 4) for sample sets collected at individual locations. We then estimated the correlation coefficient and *p*-values based on the regression of the average values of DTT and chemical components at these studied sites. According to the table, the DTT activity is mostly correlated with the K⁺ ($R = 0.94$), a marker of biomass burning reported in the literature.^{27,66,77} The observed regression is also statistically significant ($p = 0.06$), corroborating the robust redox activity of this emission source. Strong and significant ($p = 0.08$) correlation between WSOC mass fraction and DTT activities ($R = 0.89$) was also observed. This observation agrees with previous studies reporting WSOC as one of the driving factors in the DTT consumption rate.^{26,32} The highest WSOC

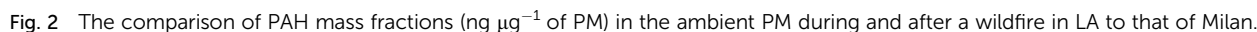
content was measured at Athens with the mass fraction of $147 \mu\text{g mg}^{-1}$ PM, followed by Milan ($\sim 118 \mu\text{g mg}^{-1}$ PM), Beirut ($\sim 52 \mu\text{g mg}^{-1}$ PM) and Riyadh during non-dust ($\sim 44 \mu\text{g mg}^{-1}$ PM) and dust periods ($\sim 13 \mu\text{g mg}^{-1}$ PM). WSOC is mainly formed by biomass burning and secondary photochemical reactions,^{78–80} however the formation of WSOC *via* photochemical reactions in winter-time is limited by the low temperatures. Thus, the large fraction of WSOC in Milan, which exhibited highest DTT activity among all locations, is most likely associated with the domestic biomass burning activities in this region. The water-insoluble organic carbon (WIOC) content, calculated as the difference between OC and WSOC, was also significantly correlated with the DTT activity ($R = 0.71$, $p = 0.02$). Since the DTT analysis was performed on the extracted filters, driven by water-soluble compounds, the high correlation of insoluble fraction of organic compounds with DTT values can be attributed to the fact that WIOC is likely emitted from similar sources as water-soluble DTT-active compounds. The PAHs have also clearly contributed to the overall redox activity of PM as corroborated by the regression values ($R = 0.74$, $p = 0.03$). PAH exposure can

Table 5 The regression analysis between PM constituents and intrinsic DTT activity ($N = 6$)^a

PM constituents	<i>R</i>	<i>p</i>
EC	0.57	0.31
OC	0.81	0.02
WSOC	0.89	0.09
WIOC	0.71	0.02
NO ₃ ⁻	0.61	0.28
SO ₄ ²⁻	0.29	0.12
K ⁺	0.95	0.06
PAHs	0.74	0.03
Ca	-0.50	0.06
Al	-0.54	0.02
Fe	-0.39	0.01
Zn	-0.76	0.02
Ba	-0.51	0.09
Cu	-0.56	0.06
Ti	-0.56	0.04
Mn	-0.67	0.02
Pb	-0.01	0.01
Ni	-0.47	0.16
Cr	0.09	0.00
V	-0.08	0.24

^a The correlation coefficient values above 0.7 and the *p*-values below 0.1 are highlighted.





The heavy-trafficked city of LA showed moderate redox activity (28.10 ± 5.23 nmoles (min mg)⁻¹) which is consistent with the moderate correlation of EC, a marker of vehicle exhaust emission, with observed DTT levels ($R = 0.57$), as well as with the high and significant correlation between DTT and WIOC ($R = 0.89$), with the latter species being a marker of primary OC emissions mainly from traffic.⁷⁸ Metals, which are typically emitted from vehicular abrasion and combustion sources (*e.g.*, industrial and traffic emissions),^{35,51,73} are reported as one of the contributors to the PM oxidative potential.^{59,84} Specifically, transition metals have been shown to have a large oxidative capacity.^{3,16} However, the DTT consumption rates across the sites chosen in our study, were not correlated with the corresponding content of metals, as evident by negative correlation coefficient for nearly all metal species. This observation could be due to two prime reasons. First, the greater potency of WSOC and OC species on a per PM mass basis in generating redox activity compared to PM elemental content may have led to negative R values between DTT and the mass fraction of metal components. PM samples with high metals and low WSOC and OC contents had the lowest DTT responses. For example, the total metallic mass fraction in Riyadh was the highest among the studied cities during both dust (~ 299 $\mu\text{g mg}^{-1}$ PM) and non-dust periods (~ 300 $\mu\text{g mg}^{-1}$ PM), while the WSOC per mass content was the lowest, resulting in lowest intrinsic DTT values during dust (9.32 ± 0.80 nmoles (min mg)⁻¹) and non-dust periods (12.53 ± 1.43 nmoles (min mg)⁻¹) in this Middle Eastern city. Secondly, the water-solubility and oxidation state of the PM components play a major role in their oxidation

activity. The consumption of the DTT assay used on the filters is primarily driven by water-soluble compounds. Therefore, the lack of correlation between DTT consumption rate and the PM total metal content, can also be attributed to the water-insoluble fraction of some metals that decreases their correlation with water-soluble DTT values.

In summary, six sets of PM samples, each dominated by unique emission sources in various locations around the globe were collected and analyzed for their content of chemical constituents and the associated oxidative potential. The DTT acellular assay was employed to quantify the redox activity of PM samples. According to our results, the highest DTT activity were observed at locations where PM were rich in OC, WSOC and PAHs, indicating that PM emitted by biomass burning activities as well as secondary organic aerosols formed by photochemical reactions have a higher oxidative potential.

Conflicts of interest

There are no conflicts of interest to declare.

Acknowledgements

This study was supported by the National Institutes of Health (NIH) (grants number: P01AG055367-03). The authors wish to thank the financial support from University of Southern California PhD fellowship award.

References

- 1 J. S. Apte, J. D. Marshall, A. J. Cohen and M. Brauer, Addressing global mortality from ambient PM_{2.5}, *Environ. Sci. Technol.*, 2015, **49**(13), 8057–8066.
- 2 A. J. Cohen, M. Brauer, R. Burnett, H. R. Anderson, J. Frostad, K. Estep, K. Balakrishnan, B. Brunekreef, L. Dandona, R. Dandona and V. Feigin, Estimates and 25-year trends of the global burden of disease attributable to ambient air pollution: an analysis of data from the Global Burden of Diseases Study 2015, *Lancet*, 2017, **389**(10082), 1907–1918.
- 3 K. R. Daellenbach, G. Uzu, J. Jiang, L. E. Cassagnes, Z. Leni, A. Vlachou, G. Stefenelli, F. Canonaco, S. Weber, A. Segers and J. J. Kuenen, Sources of particulate-matter air pollution and its oxidative potential in Europe, *Nature*, 2020, **587**(7834), 414–419.
- 4 R. J. Delfino, C. Sioutas and S. Malik, Potential role of ultrafine particles in associations between airborne particle mass and cardiovascular health, *Environ. Health Perspect.*, 2005, **113**(8), 934–946.
- 5 Y. Du, X. Xu, M. Chu, Y. Guo and J. Wang, Air particulate matter and cardiovascular disease: the epidemiological, biomedical and clinical evidence, *J. Thorac. Dis.*, 2016, **8**(1), E8.
- 6 G. Grande, P. L. Ljungman, K. Eneroth, T. Bellander and D. Rizzuto, Association between cardiovascular disease and long-term exposure to air pollution with the risk of dementia, *JAMA Neurol.*, 2020, **77**(7), 801–809.
- 7 K. Shrey, A. Suchit, D. Deepika, K. Shruti and R. Vibha, Air pollutants: the key stages in the pathway towards the development of cardiovascular disorders, *Environ. Toxicol. Pharmacol.*, 2011, **31**(1), 1–9.
- 8 N. A. Potter, G. Y. Meltzer, O. N. Avenbuan, A. Raja and J. T. Zelikoff, Particulate matter and associated metals: A link with neurotoxicity and mental health, *Atmosphere*, 2021, **12**(4), 425.
- 9 S. Roberts, L. Arseneault, B. Barratt, S. Beevers, A. Danese, C. L. Odgers, T. E. Moffitt, A. Reuben, F. J. Kelly and H. L. Fisher, Exploration of NO₂ and PM_{2.5} air pollution and mental health problems using high-resolution data in London-based children from a UK longitudinal cohort study, *Psychiatr. Res.*, 2019, **272**, 8–17.
- 10 N. C. Woodward, A. Haghani, R. G. Johnson, T. M. Hsu, A. Saffari, C. Sioutas, S. E. Kanoski, C. E. Finch and T. E. Morgan, Prenatal and early life exposure to air pollution induced hippocampal vascular leakage and impaired neurogenesis in association with behavioral deficits, *Transl. Psychiatry*, 2018, **8**(1), 1–10.
- 11 B. J. Lee, B. Kim and K. Lee, Air pollution exposure and cardiovascular disease, *Toxicol. Res.*, 2014, **30**(2), 71–75.
- 12 E. J. Jo, W. S. Lee, H. Y. Jo, C. H. Kim, J. S. Eom, J. H. Mok, M. H. Kim, K. Lee, K. U. Kim, M. K. Lee and H. K. Park, Effects of particulate matter on respiratory disease and the impact of meteorological factors in Busan, Korea, *Respir. Med.*, 2017, **124**, 79–87.
- 13 J. Li, W. X. Li, C. Bai and Y. Song, Particulate matter-induced epigenetic changes and lung cancer, *Clin. Respir. J.*, 2017, **11**(5), 539–546.
- 14 J. B. Soriano, P. J. Kendrick, K. R. Paulson, V. Gupta, E. M. Abrams, R. A. Adedoyin, T. B. Adhikari, S. M. Advani, A. Agrawal, E. Ahmadian and F. Alahdab, Prevalence and attributable health burden of chronic respiratory diseases, 1990–2017: a systematic analysis for the Global Burden of Disease Study 2017, *Lancet Respir. Med.*, 2020, **8**(6), 585–596.
- 15 R. Tohidi, B. Sajadi and G. Ahmadi, The effect of nasal airway obstruction on the dispersion and deposition of inhaled volatile droplets in the human nasal cavity: A numerical study, *J. Aerosol Sci.*, 2020, **150**, 105650.
- 16 J. T. Bates, T. Fang, V. Verma, L. Zeng, R. J. Weber, P. E. Tolbert, J. Y. Abrams, S. E. Sarnat, M. Klein, J. A. Mulholland and A. G. Russell, Review of acellular assays of ambient particulate matter oxidative potential: methods and relationships with composition, sources, and health effects, *Environ. Sci. Technol.*, 2019, **53**(8), 4003–4019.
- 17 P. P. Fu, Q. Xia, H. M. Hwang, P. C. Ray and H. Yu, Mechanisms of nanotoxicity: generation of reactive oxygen species, *J. Food Drug Anal.*, 2014, **22**(1), 64–75.
- 18 A. Fushimi, D. Nakajima, A. Furuyama, G. Suzuki, T. Ito, K. Sato, Y. Fujitani, Y. Kondo, A. Yoshino, S. Ramasamy and J. J. Schauer, Source contributions to multiple toxic potentials of atmospheric organic aerosols, *Sci. Total Environ.*, 2021, **773**, 145614.
- 19 E. Paszti-Gere, E. Csibrik-Nemeth, K. Szeker, R. Csizinszky, C. Jakab and P. Galfi, Acute oxidative stress affects IL-8



- and TNF- α expression in IPEC-J2 porcine epithelial cells, *Inflammation*, 2012, **35**(3), 994–1004.
- 20 M. Song, S. H. Oh, C. Park and M. S. Bae, Analytical Procedure for Dithiothreitol-based Oxidative Potential of PM 2.5, *Asian J. Atmos. Environ.*, 2021, **15**(1), 1–9.
 - 21 F. Tao, B. Gonzalez-Flecha and L. Kobzik, Reactive oxygen species in pulmonary inflammation by ambient particulates, *Free Radic. Biol. Med.*, 2003, **35**(4), 327–340.
 - 22 S. Weichenthal, E. Lavigne, G. Evans, K. Pollitt and R. T. Burnett, Ambient PM2.5 and risk of emergency room visits for myocardial infarction: impact of regional PM2.5 oxidative potential: a case-crossover study, *Environ. Health*, 2016, **15**(1), 1–9.
 - 23 A. K. Cho, C. Sioutas, A. H. Miguel, Y. Kumagai, D. A. Schmitz, M. Singh, A. Eiguren-Fernandez and J. R. Froines, Redox activity of airborne particulate matter at different sites in the Los Angeles Basin, *Environ. Res.*, 2005, **99**(1), 40–47.
 - 24 A. Saffari, N. Daher, M. M. Shafer, J. J. Schauer and C. Sioutas, Global perspective on the oxidative potential of airborne particulate matter: a synthesis of research findings, *Environ. Sci. Technol.*, 2014a, **48**(13), 7576–7583.
 - 25 V. Verma, R. Rico-Martinez, N. Kotra, L. King, J. Liu, T. W. Snell and R. J. Weber, Contribution of water-soluble and insoluble components and their hydrophobic/hydrophilic subfractions to the reactive oxygen species-generating potential of fine ambient aerosols, *Environ. Sci. Technol.*, 2012, **46**(20), 11384–11392.
 - 26 H. Vreeland, R. Weber, M. Bergin, R. Greenwald, R. Golan, A. G. Russell, V. Verma and J. A. Sarnat, Oxidative potential of PM2.5 during Atlanta rush hour: Measurements of in-vehicle dithiothreitol (DTT) activity, *Atmos. Environ.*, 2017, **165**, 169–178.
 - 27 Y. Cheng, G. Engling, K. B. He, F. K. Duan, Y. L. Ma, Z. Y. Du, J. M. Liu, M. Zheng and R. J. Weber, Biomass burning contribution to Beijing aerosol, *Atmos. Chem. Phys.*, 2013, **13**(15), 7765–7781.
 - 28 N. Daher, N. A. Saliba, A. L. Shihadeh, M. Jaafar, R. Baalbaki, M. M. Shafer, J. J. Schauer and C. Sioutas, Oxidative potential and chemical speciation of size-resolved particulate matter (PM) at near-freeway and urban background sites in the greater Beirut area, *Sci. Total Environ.*, 2014, **470**, 417–426.
 - 29 H. J. Kim, M. G. Choi, M. K. Park and Y. R. Seo, Predictive and prognostic biomarkers of respiratory diseases due to particulate matter exposure, *J. Cancer Prev.*, 2017, **22**(1), 6.
 - 30 H. Ma, J. Li, C. Wan, Y. Liang, X. Zhang, G. Dong, L. Hu, B. Yang, X. Zeng, T. Su and S. Lu, Inflammation response of water-soluble fractions in atmospheric fine particulates: a seasonal observation in 10 large Chinese cities, *Environ. Sci. Technol.*, 2019, **53**(7), 3782–3790.
 - 31 A. Saffari, N. Daher, M. M. Shafer, J. J. Schauer and C. Sioutas, Seasonal and spatial variation in dithiothreitol (DTT) activity of quasi-ultrafine particles in the Los Angeles Basin and its association with chemical species, *J. Environ. Sci. Health, Part A: Toxic/Hazard. Subst. Environ. Eng.*, 2014b, **49**(4), 441–451.
 - 32 V. Verma, Z. Ning, A. K. Cho, J. J. Schauer, M. M. Shafer and C. Sioutas, Redox activity of urban quasi-ultrafine particles from primary and secondary sources, *Atmos. Environ.*, 2009a, **43**(40), 6360–6368.
 - 33 J. Z. Wu, D. D. Ge, L. F. Zhou, L. Y. Hou, Y. Zhou and Q. Y. Li, Effects of particulate matter on allergic respiratory diseases, *Chronic Dis. Transl. Med.*, 2018, **4**(02), 95–102.
 - 34 C. Lovett, M. H. Sowlat, N. A. Saliba, A. L. Shihadeh and C. Sioutas, Oxidative potential of ambient particulate matter in Beirut during Saharan and Arabian dust events, *Atmos. Environ.*, 2018, **188**, 34–42.
 - 35 V. J. Farahani, E. Soleimanian, M. Pirhadi and C. Sioutas, Long-term trends in concentrations and sources of PM2.5-bound metals and elements in central Los Angeles, *Atmos. Environ.*, 2021, **253**, 118361.
 - 36 M. Paglione, S. Gilardoni, M. Rinaldi, S. Decesari, N. Zanca, S. Sandrini, L. Giulianelli, D. Bacco, S. Ferrari, V. Poluzzi and F. Scotto, The impact of biomass burning and aqueous-phase processing on air quality: a multi-year source apportionment study in the Po Valley, Italy, *Atmos. Chem. Phys.*, 2020, **20**(3), 1233–1254.
 - 37 R. German, H. Nijland, A. Pridmore, C. Ahlgren and T. Williamson, Vehicle Emissions and Impacts of Taxes and Incentives in the Evolution of Past Emissions, *Report to the EEA, ETC/ACM*, 2018, vol. 1.
 - 38 K. Eleftheriadis, Long-term variability of the air pollution sources reflected on the state of mixing of the urban aerosol, *Curr. Opin. Environ. Sci. Health*, 2019, **8**, 36–39.
 - 39 A. Baayoun, W. Itani, J. El Helou, L. Halabi, S. Medlej, M. El Malki, A. Moukhadder, L. K. Aboujaoude, V. Kabakian, H. Mounajed and T. Mokalled, Emission inventory of key sources of air pollution in Lebanon, *Atmos. Environ.*, 2019, **215**, 116871.
 - 40 W. Marrouch and J. Mourad, Effect of gasoline prices on car fuel efficiency: Evidence from Lebanon, *Energy Pol.*, 2019, **135**, 111001.
 - 41 M. Hakimzadeh, E. Soleimanian, A. Mousavi, A. Borgini, C. De Marco, A. A. Ruprecht and C. Sioutas, The impact of biomass burning on the oxidative potential of PM2.5 in the metropolitan area of Milan, *Atmos. Environ.*, 2020, **224**, 117328.
 - 42 M. E. Birch and R. A. Cary, Elemental carbon-based method for monitoring occupational exposures to particulate diesel exhaust, *Aerosol Sci. Technol.*, 1996, **25**(3), 221–241.
 - 43 E. A. Stone, D. C. Snyder, R. J. Sheesley, A. P. Sullivan, R. J. Weber and J. J. Schauer, Source apportionment of fine organic aerosol in Mexico City during the MILAGRO experiment 2006, *Atmos. Chem. Phys.*, 2008, **8**(5), 1249–1259.
 - 44 G. Drago, C. Perrino, S. Canepari, S. Ruggieri, L. L'Abbate, V. Longo, P. Colombo, D. Frasca, M. Balzan, G. Cuttitta and G. Scaccianocce, Relationship between domestic smoking and metals and rare earth elements concentration in indoor PM2.5, *Environ. Res.*, 2018, **165**, 71–80.
 - 45 G. C. Lough, J. J. Schauer, J. S. Park, M. M. Shafer, J. T. DeMinter and J. P. Weinstein, Emissions of metals



- This article is licensed under a Creative Commons Attribution 3.0 Unported Licence.



- Elements and polycyclic aromatic hydrocarbons in exhaust particles emitted by light-duty vehicles, *Environ. Sci. Pollut. Res.*, 2015, **22**(15), 11526–11542.
- 69 E. Galarneau, Source specificity and atmospheric processing of airborne PAHs: implications for source apportionment, *Atmos. Environ.*, 2008, **42**(35), 8139–8149.
- 70 H. Guo, S. C. Lee, K. F. Ho, X. M. Wang and S. C. Zou, Particle-associated polycyclic aromatic hydrocarbons in urban air of Hong Kong, *Atmos. Environ.*, 2003, **37**(38), 5307–5317.
- 71 A. L. Lima, J. W. Farrington and C. M. Reddy, Combustion-derived polycyclic aromatic hydrocarbons in the environment—a review, *Environ. Forensics*, 2005, **6**(2), 109–131.
- 72 L. Miller, L. D. Lemke, X. Xu, S. M. Molaroni, H. You, A. J. Wheeler, J. Booza, A. Grgicak-Mannion, R. Krajenta, P. Graniero and H. Krouse, Intra-urban correlation and spatial variability of air toxics across an international airshed in Detroit, Michigan (USA) and Windsor, Ontario (Canada), *Atmos. Environ.*, 2010, **44**(9), 1162–1174.
- 73 F. Shirmohammadi, D. Wang, S. Hasheminassab, V. Verma, J. J. Schauer, M. M. Shafer and C. Sioutas, Oxidative potential of on-road fine particulate matter (PM_{2.5}) measured on major freeways of Los Angeles, CA, and a 10-year comparison with earlier roadside studies, *Atmos. Environ.*, 2017, **148**, 102–114.
- 74 S. O. Baek, R. A. Field, M. E. Goldstone, P. W. Kirk, J. N. Lester and R. Perry, A review of atmospheric polycyclic aromatic hydrocarbons: sources, fate and behavior, *Water Air Soil Pollut.*, 1991, **60**(3), 279–300.
- 75 D. Grosjean, K. Fung and J. Harrison, Interactions of polycyclic aromatic hydrocarbons with atmospheric pollutants, *Environ. Sci. Technol.*, 1983, **17**(11), 673–679.
- 76 K. Saarnio, M. Sillanpää, R. Hillamo, E. Sandell, A. S. Pennanen and R. O. Salonen, Polycyclic aromatic hydrocarbons in size-segregated particulate matter from six urban sites in Europe, *Atmos. Environ.*, 2008, **42**(40), 9087–9097.
- 77 C. Reche, M. Viana, F. Amato, A. Alastuey, T. Moreno, R. Hillamo, K. Teinilä, K. Saarnio, R. Seco, J. Peñuelas and C. Mohr, Biomass burning contributions to urban aerosols in a coastal Mediterranean City, *Sci. Total Environ.*, 2012, **427**, 175–190.
- 78 S. Hasheminassab, N. Daher, J. J. Schauer and C. Sioutas, Source apportionment and organic compound characterization of ambient ultrafine particulate matter (PM) in the Los Angeles Basin, *Atmos. Environ.*, 2013, **79**, 529–539.
- 79 R. J. Weber, A. P. Sullivan, R. E. Peltier, A. Russell, B. Yan, M. Zheng, J. De Gouw, C. Warneke, C. Brock, J. S. Holloway and E. L. Atlas, A study of secondary organic aerosol formation in the anthropogenic-influenced southeastern United States, *J. Geophys. Res.: Atmos.*, 2007, **16**(D13), 112.
- 80 X. Zhang, Z. Liu, A. Hecobian, M. Zheng, N. H. Frank, E. S. Edgerton and R. J. Weber, Spatial and seasonal variations of fine particle water-soluble organic carbon (WSOC) over the southeastern United States: implications for secondary organic aerosol formation, *Atmos. Chem. Phys.*, 2012, **12**(14), 6593–6607.
- 81 N. Podechard, V. Lecureur, E. Le Ferrec, I. Guenon, L. Sparfel, D. Gilot, J. R. Gordon, V. Lagente and O. Fardel, Interleukin-8 induction by the environmental contaminant benzo (a) pyrene is aryl hydrocarbon receptor-dependent and leads to lung inflammation, *Toxicol. Lett.*, 2008, **177**(2), 130–137.
- 82 C. Gutiérrez-Vázquez and F. J. Quintana, Regulation of the immune response by the aryl hydrocarbon receptor, *Immunity*, 2018, **48**(1), 19–33.
- 83 S. Taghvaei, M. H. Sowlat, E. Diapouli, M. I. Manousakas, V. Vasilatou, K. Eleftheriadis and C. Sioutas, Source apportionment of the oxidative potential of fine ambient particulate matter (PM_{2.5}) in Athens, Greece, *Sci. Total Environ.*, 2019, **653**, 1407–1416.
- 84 D. Gao, S. Ripley, S. Weichenthal and K. J. Pollitt, Ambient particulate matter oxidative potential: Chemical determinants, associated health effects, and strategies for risk management, *Free Radic. Biol. Med.*, 2020, **151**, 7–25.
- 85 M. Pirhadi, A. Mousavi, S. Taghvaei, M. M. Shafer and C. Sioutas, Semi-volatile components of PM_{2.5} in an urban environment: volatility profiles and associated oxidative potential, *Atmos. Environ.*, 2020, **223**, 117197.
- 86 A. Altuwayjiri, M. Pirhadi, M. Kalafy, B. Alharbi and C. Sioutas, Impact of different sources on the oxidative potential of ambient particulate matter PM₁₀ in Riyadh, Saudi Arabia: A focus on dust emissions, *Sci. Total Environ.*, 2022, **806**, 150590.