Review

Estimation of the contribution of road traffic emissions to particulate matter concentrations from field measurements: A review

CrossMark

Pallavi Pant, Roy M. Harrison[*],[1]

Division of Environmental Health & Risk Management, School of Geography, Earth & Environmental Sciences, University of Birmingham, Edgbaston, Birmingham B15 2TT, United Kingdom

HIGHLIGHTS

- Road traffic contributes emissions from exhaust, abrasion and re-suspension sources.
- Chemical and physical properties of the emitted particles are described.
- Available methods for quantification of traffic-derived concentrations are reviewed.

ARTICLE INFO

ABSTRACT

Road traffic is one of the main sources of particulate matter in the atmosphere. Despite its importance, there are significant challenges in quantitative evaluation of its contribution to airborne concentrations. This article first reviews the nature of the particle emissions from road vehicles including both exhaust and non-exhaust (abrasion and re-suspension sources). It then briefly reviews the various methods available for quantification of the road traffic contribution. This includes tunnel/roadway measurements, twin site studies, use of vehicle-specific tracers and other methods. Finally, the application of receptor modelling methods is briefly described. Based on the review, it can be concluded that while traffic emissions continue to contribute substantially to primary PM emissions in urban areas, quantitative knowledge of the contribution, especially of non-exhaust emissions to PM concentrations remain inadequate.

© 2013 Elsevier Ltd. All rights reserved.

1. Introduction

Emissions due to road traffic are known to make a large contribution to total particulate matter (PM) concentrations in urban areas and exposure to PM from vehicular emissions has been demonstrated to have detrimental impacts on human health (Mauderly, 1994; Buckeridge et al., 2002; Fan et al., 2006; HEI, 2010; Masiol et al., 2012; Rissler et al., 2012). An improved understanding of the characteristics of emissions from different traffic-related sources is therefore vital for conducting source apportionment and health effects studies (Schauer et al., 2006). PM emissions from road vehicles include emissions from the tailpipe (exhaust emissions) and emissions due to wear and tear of vehicle parts such as brake, tyre and clutch and re-suspension of dust (non-exhaust

emissions). Non-exhaust emissions contribute mainly to the coarse mode of PM ($PM_{2.5-10}$) while exhaust emissions contribute predominantly to fine PM (aerodynamic diameter < 2.5 μm) (Abu-Allaban et al., 2003; Tervahattu et al., 2006; Thorpe et al., 2007; Kam et al., 2012) but this is a far from clear distinction. A number of different properties of traffic emissions are studied including physical shape and structure, particle size distributions, chemical composition, and temporal and spatial variation (Watson and Chow, 2007). Most of the research as well as policy action in the last few decades has largely focused on exhaust emissions, and stringent regulations and technological upgrades have resulted in a decline of the percentage contribution of vehicle tailpipe emissions to total ambient PM concentrations (Allen et al., 2006; Thorpe and Harrison, 2008; Mathissen et al., 2011). As a result, the contribution of non-exhaust PM is becoming more important although detailed information on non-exhaust PM emissions is relatively scarce (Kuhlbusch et al., 2009; Amato et al., 2012; Harrison et al., 2011, 2012b; Denier van der Gon et al., 2013). It has been shown that even with zero tail pipe emissions, traffic will continue to contribute to fine and ultrafine particles through non-exhaust emissions (Dahl

et al., 2006; Kumar et al., 2013) and it is estimated that nearly 90% of the total emissions from road traffic will come from non-exhaust sources by the end of the decade (Rexeis and Hausberger, 2009).

In order to understand the contribution of road traffic emissions to the environment, it is important to understand various aspects including sources, chemical characteristics and quantitative contributions. In the last three decades, a significant amount of research has been conducted on characterization and analysis of exhaust emissions and as a result, there is a very strong body of knowledge in the field. Non-exhaust emissions, however, are becoming more important now, and further research is anticipated in this field in the coming years. This review aims to reconcile the various studies that have been reported around this issue, and tease out some common characteristics, and highlight areas with future research potential.

This paper reviews the current information on primary road traffic emissions including exhaust and non-exhaust emissions. Key characteristics of the major sources are discussed. Various methodologies used for measurement and analysis of such emissions are also discussed with a brief discussion on the merits and shortcomings of each method. In the final section, various source apportionment techniques useful for the estimation of traffic source contributions are discussed together with a summary of some of the reported studies. A number of in-depth review articles have been published in the last few years on various issues related to road traffic emissions on topics such as properties and characteristics of non-exhaust emissions (Thorpe and Harrison, 2008), toxicological impacts of tyre wear emissions (Wik and Dave, 2009), chemical composition of brake pads (Chan and Stachowiak, 2004), development of emission factors (Franco et al., 2013), nanoparticle emissions from non-exhaust sources (Kumar et al., 2013). However, each of these reviews focuses on one or a few specific aspects of road traffic emissions. This review builds on the information provided in those review articles, and repetition is avoided. In summary, the key questions that we hope to address through this review article are as follows:

1. What are the key characteristics/sources/contributions of road traffic emissions? [What and how much do we know about each of the road traffic emission sources?]
2. How can such emissions be characterized? [Various methods, pros and cons of the methods]
3. What have we learnt so far?
4. What are the key issues that are not resolved yet?

The literature search for this review was primarily performed using Web of Knowledge, Science Direct and Google Scholar using various combinations of keywords such as traffic emissions, non-exhaust/exhaust emissions, re-suspension, road dust, traffic PM etc. However, the authors acknowledge that not all articles published on the topic have been included in the review.

2. Non-exhaust emissions

Non-exhaust PM comprises the various emissions that do not derive from the tailpipe of a vehicle including particles generated due to brake and tyre wear, road surface abrasion, wear and tear/corrosion of other vehicle components such as the clutch, and re-suspension of road surface dusts (Table 1). Key reasons for needing to understand non-exhaust emissions include their inherent toxicity including their tendency to act as carriers of heavy metals and carcinogenic components (Adachi and Tainosho, 2004; Hjortenkrans et al., 2007; Johansson et al., 2009; Amato et al., 2011a) and their contribution to exceedances of air quality guidelines and standards (Amato et al., 2011a; Denier van der Gon et al., 2013).

Harrison et al. (2012b) showed that the size distributions of trace metals were indicative of particle sources. Keuken et al. (2010) concluded that re-suspension of accumulated PM and road wear related particles are the primary contributors to non-exhaust emissions, and tyre wear and brake wear contribute to the fine and coarse fraction respectively. However, Narvaez et al. (2008) reported that although a majority of abrasion particles are in the coarse fraction, abrasion can contribute significantly to the fine fraction of PM. In Taiwan, EFs for elements such as Ba and Se were found to be higher in the $PM_{2.5-10}$ fraction indicating mechanical friction and brake wear as potential sources (Chiang and Huang, 2009). Road traffic was found to be a key contributor to fine/ultrafine and nano mode particles of Ba, Zn and Pb (Lin et al., 2005) and in Barcelona (Spain), 62–96% of elements including Cu, Sb, Ba, Mn and Zn were found to be present in the PM_{1-10} fraction (Perez et al., 2010). A detailed review of the nanoparticle emissions from non-exhaust sources including traffic and non-traffic sources is presented in Kumar et al. (2013).

Non-exhaust emissions are typically characterized by trace metals (e.g. Cu, Zn, Ba, Sb, Mn) although organic markers (e.g. polycyclic aromatic hydrocarbons (PAHs), *n*-alkanes) have been used in some cases (Wahlin et al., 2006; Amato et al., 2009; El Haddad et al., 2009; Gietl et al., 2010; Oliveira et al., 2011; Kwon and Castaldi, 2012). A list of key tracers used for source characterization of non-exhaust emissions is presented in Table 2. However, emissions of trace metal markers are reported to vary with the fleet composition, with higher emissions reported for some of the elements for heavy duty vehicles (HDVs) (Grieshop et al., 2006; Mancilla and Mendoza, 2012). In addition, the profile of trace metal concentrations in non-exhaust particulate matter is unique for every region and varies based on parameters such as traffic volume and pattern, vehicle fleet characteristics, driving and traffic patterns and climate and geology of the region (Omstedt et al., 2005; Amato et al., 2011a,b; Duong and Lee, 2011). Another important aspect is the variability of tyre and brake composition depending on the manufacturer which makes it very difficult to ascertain fleet-wide composition other than from environmental measurements (Canepari et al., 2008; Gietl et al., 2010; Denier van der Gon et al., 2013).

Several factors are reported to affect non-exhaust emissions including increase in vehicle speed (Chen et al., 2006; Gustafsson et al., 2008; Hussein et al., 2008; Mathissen et al., 2011) although Keuken et al. (2010) reported a non-linear relationship between traffic volume and non-exhaust emissions. Metal emissions due to road dust show a low correlation to metal emissions due to abrasion/brake wear and combustion (Johansson et al., 2009). One of the major problems in analysis of non-exhaust PM using field data has been the difficulty in distinguishing between wear and tear emissions and road dust since the chemical composition is often very similar (Bukowiecki et al., 2010). This may, in any case, not be a

Table 1
Mechanisms for non-exhaust particle emission (Rogge et al., 1993; Boulter et al., 2005; Wahlin et al., 2006; Thorpe et al., 2007; Amato et al., 2009; Harrison, 2009; Gietl et al., 2010; Barmpadimos et al., 2011; Hays et al., 2011; Denier van der Gon et al., 2013).

Emission type	Mechanism	Includes
Direct	Abrasion and wear and tear	Tyre, brake, clutch, road surface
	Corrosion	Vehicle, street furniture
Indirect	Re-suspension (due to tyre shear, wind and vehicle turbulence)	

Table 2
Key tracers used for non-exhaust PM.

Reference	Brake wear	Tyre wear	Re-suspension
Adachi and Tainosho (2004)	Fe, Ba, Cu, Sb, Zr	Zn	–
Schauer et al. (2006)	Fe, Cu, Ba		–
Grieshop et al. (2006)	Cu, Sb, Ba and Ga		
Wahlin et al. (2006)	Cr, Fe, Cu, Zn, Zr, Mo, Sn, Sb, Ba and Pb	Al, Si, K, Ca, Ti, Mn, Fe, Zn and Sr (together with road dust)	
Tanner et al. (2008)	Cu, Cd	Zn	
Canepari et al. (2008)	Ba, Fe, Sb, Sr	–	–
Harrison (2009)	Ba, Cu	–	Al
Dongarra et al. (2009)	Cu, Mo, Sb	–	
Fabretti et al. (2009)	Cu, Zn, Sb, Sn (vehicular abrasion)	–	Rb, Sr, Mn, Fe, As
Keuken et al. (2010)	Cu	Zn	–
Bukowiecki et al. (2010)	Fe, Cu, Zn, Zr, Mo, Sn, Sb and Ba		
Pey et al. (2010)	–	–	Fe, Ca, Sb, Sn, Cu, Zn
Perez et al. (2010)	Sb, Cu, Ni, Sn (wear of brake, tire and other parts)		Fe
Amato et al. (2011a)	Fe, Cu, Zn, Cr, Sn, Sb	OC, S, Zn	Al, Ca, Fe, V
Apeagyei et al. (2011)	Fe, Ti, Cu, Ba	Zn, Ca, W, K, Fe, Ti, Cr, Mo	–
Duong and Lee (2011)	Ni, Cu	Zn	
Ondráček et al. (2011)	Cu, Ba, Fe, Zn	–	
Song and Gao (2011)	Sb, Cu, Fe, Pb	Zn, Co	
Sahu et al. (2011)	Zn (brake and tyre wear)	–	
Peltier et al. (2011)	–	–	Al, Si, Ti, Fe
Harrison et al. (2012b)	Ba, Cu, Fe, Sb	Zn	Si, Al

clear distinction as wear emissions may deposit to the road surface, only to be re-suspended subsequently. Gehrig et al. (2010) used mobile load simulators to compare the contributions of road abrasion and re-suspension emissions, and concluded that particle emissions due to abrasion are a function of the state of the pavement whereas re-suspension is normally higher, and increases with higher dust loads on the road. Thorpe and Harrison (2008) present a more elaborate account of sources and properties of non-exhaust emissions.

Mixed results have been obtained for the impact of street washing activities on PM and while Amato et al. (2009) observed a reduction, albeit statistically non-significant, in the concentration of trace metals known to be contributed by non-exhaust traffic sources in Spain, Keuken et al. (2010) reported a lack of significant reduction in the non-exhaust emissions after street sweeping or washing in the Netherlands.

2.1. Tyre wear

Tyre wear particles are generated either by shear forces between the tyre tread and the road surface and are predominantly coarse ($PM_{2.5–10}$) or by volatilization (Rogge et al., 1993; Kupiainen et al., 2005; Allen et al., 2006; Thorpe and Harrison, 2008; Aatmeeyata and Sharma, 2010). Tyre wear particles are reported to be generally elongated with rough surfaces based on microscopic analysis (Gunawardana et al., 2011). Tyre tread, a source of airborne particles, contains natural rubber copolymers such as styrene–butadiene rubber and polyisoprene rubber, and zinc (Zn) is added to tyre tread as zinc oxide and organozinc compounds to facilitate the vulcanization process. Passenger car tyres in EU are known to contain nearly 1% zinc oxide, and rubber, metals and carbon black make up typically 47%, 16.5% and 21.5% respectively (Milani et al., 2004). The key tracer components of tyre wear include n-alkanes, n-alkanoic acids, PAHs, benzothiazoles and trace metals (Rogge et al., 1993; Reddy and Quinn, 1997; Camatani et al., 2001; Adachi and Tainosho, 2004; Boulter et al., 2005; Aatmeeyata and Sharma, 2010). Zn is reported to be nearly 1% by weight in rubber tyres (Councell et al., 2004) and tyre wear has been reported to be a significant source of Zn (Adachi and Tainosho, 2004; Hjortenkrans et al., 2007; Ondráček et al., 2011). The concentration of Zn was found to be approximately 15 times higher in tyres compared to brakes while concentrations of other heavy metals such as copper

(Cu) and barium (Ba) were higher for brake materials in Massachusetts (USA) (Apeagyei et al., 2011). Differences have also been observed between tyre tread composition (primarily Zn) and tyre wear (Al, Si, Ca, Fe, Zn, Ti) (Adachi and Tainosho, 2004). However, the spatial and temporal trend for Zn was found to be different from other roadside tracers in New York (USA), and a very small portion of airborne Zn concentration was attributed to tyre wear (Peltier et al., 2011). However, Zn is emitted from brake wear, motor oil and other sources and cannot be used as the only tracer for tyre wear (Lough et al., 2005). Benzothiazoles are also used as markers for tyre wear, particularly benzothiazole (BT), 2-hydroxy benzothiazole (HOBT), 2-(4-morpholinyl)benzothiazole (24MoBT) and N-cyclohexyl-2-benzothiazolamine (NCBA) (Kumata et al., 2002; Allen et al., 2006; Wik and Dave, 2009). Among the PAHs, pyrene, benzo(ghi)perylene, fluoranthene and phenanthrene are known to be emitted from tyres (Boonyatumanond et al., 2007; Kwon and Castaldi, 2012). Aatmeeyata and Sharma (2010) reported that small cars tested in India emitted 378 ng of total PAH/tyre km while in the USA, 200 $\mu g\ g^{-1}$ of PAHs were found in tyre wear with pyrene, fluoranthene and phenanthrene being present in the highest concentrations (Rogge et al., 1993). Tyre tread wear has also been found to contain an average of 0.53 $\mu g\ g^{-1}$ of dibenzopyrenes, indicating that tyre dust may be significant source of dibenzopyrenes in the environment (Sadiktsis et al., 2012).

Average tread wear for tyres is reported to be between 0.006 and 0.009 g km^{-1} based on the road, tyre and vehicle conditions (Rogge et al., 1993) and tyres can lose up to 10% of their mass during their lifetime (Milani et al., 2004). Speed has been reported as an influential parameter for particle mass and number concentrations for tyre wear emissions (Gustafsson et al., 2008; Mathissen et al., 2011). Other important parameters for tyre tread wear include road surface type, and tyre and driving conditions (Allen et al., 2006; Gustafsson et al., 2008). For example, asphalt surfaces have been reported to cause less tyre wear than concrete pavements and in Arizona, USA, the emission rate for tyre wear was found to be 1.4–2 times lower for asphalt pavement compared to concrete pavement (Allen et al., 2006). The type of tyre also impacts the magnitude of tyre wear emissions and studded tyres are known to cause more emissions compared with summer and friction tyres (Kupiainen et al., 2005; Gustafsson et al., 2008; Hussein et al., 2008; Schaap et al., 2009). Unimodal (70–90 nm) and bimodal (<10 and 30–60 nm) peaks in the nano size range have been reported for

tyre particles by number under low and high speed conditions respectively (Mathissen et al., 2011).

2.2. Brake wear

Brake wear, including abrasion of brake lining material and brake discs, caused by grinding of brake pad constituents (coarse range particles) or volatilization and condensation of brake pad materials (fine range particles), is known to release PM directly into the atmosphere and to contribute to the trace metal concentration in airborne PM, particularly less than 10 μm (Garg et al., 2000; Blau and Meyer, 2003; Ingo et al., 2004; Wahlin et al., 2006; Gietl et al., 2010; Varrica et al., 2012). Key components of brake pads include fillers, frictional additives, reinforcing fibres and binder (Chan and Stachowiak, 2004) and the key chemical species used include sulphides of metal, abrasives (e.g. silica), barium silicate/sulphate (particularly in brake linings) and other metallic particles (as filler material), carbon fibres and lubricant (e.g. graphite) (Ingo et al., 2004; Dongarra et al., 2009). Chan and Stachowiak (2004) present a detailed review of the chemical constituents of brake pads. Garg et al. (2000) observed an average mass median diameter (median particle diameter based on mass) of 1.49 μm for brake wear particles using tests on a brake dynamometer whereas Sanders et al. (2003) reported a mass median diameter of 6 μm for brake debris generated during urban driving conditions. Higher brake wear related emissions have been reported during rush hour. A higher number of particles are released in the braking phase as compared to the acceleration phase (Hussein et al., 2008; Mathissen et al., 2011) and various studies have corroborated this observations including Grieshop et al. (2006) who reported a higher particle number concentration during rush hour where the traffic often operates in the stop-and-go mode and Abu-Allaban et al. (2003) who observed higher contribution of brake wear at freeway exit sites compared to other types of roadside sites (Abu-Allaban et al., 2003).

Passenger cars have been estimated to emit nearly 44 g car^{-1} year^{-1} brake dust (Iijima et al., 2007). Sanders et al. (2003) reported high concentrations of Fe, Ba and Cu in brake lining wear using dynamometer and track tests whereas Adachi and Tainosho (2004) reported Fe to be the most abundant metal in brake dust along with other metals such as Ba, Cu, Sb, Zr and Zn, and Hjortenkrans et al. (2007) reported Cu, Zn and Sb to be present in brake wear emissions. Varrica et al. (2012) undertook a detailed study on Sb in brake dust and reported the most commonly released forms of Sb due to brake abrasion to be Sb (III) and Sb (V). However, brake pads are often found to have different composition based on the brand, particularly with respect to Cu and Sb (Hjortenkrans et al., 2007; Canepari et al., 2008). Polyalkylene glycol ethers are also reported to be present in brake wear particles with small concentrations of n-alkanes and n-alkanoic acids (Rogge et al., 1993).

Iijima et al. (2007) reported a unimodal number-based distribution for brake abrasion dust with the mode at 1–2 μm while the mass-based distribution had a peak at 3–6 μm. EFs for brake wear have been reported by several studies including Garg et al. (2000) who reported a brake wear EF of 3–9 mg km^{-1} for light duty gasoline vehicles (LDGV) and Abu-Allaban et al. (2003) who reported an EF of 0–80 mg km^{-1}. Wahlin et al. (2006), on the other hand, reported a Cu emission factor of 0.7 ± 0.2 mg km^{-1} vehicle in Copenhagen.

Sternbeck et al. (2002) proposed the ratio of Cu:Sb (4.6 ± 2.3) as characteristic of brake wear particles and Table 3 lists the Cu: Sb ratios as reported in the literature. Differences in the ratios can be attributed to the difference in the brake pad composition, and contributions of metals from other sources. In some cases, concentrations in ambient air can also vary due to the site characteristics.

2.3. Road dust and road surface wear

Road dust, of which crustal dust is a key component, consists of primarily coarse-sized particles derived from different sources such as traffic, industrial emissions, mineralogical dust etc. (Kupiainen et al., 2005; Tanner et al., 2008). Composition of road dust shows spatial as well as temporal variation and it is often difficult to classify dust into crustal/re-suspended/direct emission etc. In Monterrey (Mexico), re-suspended dust was found to be contributing nearly 20–25% to the PM$_{2.5}$ EF (Mancilla and Mendoza, 2012). The amount of re-suspended road dust particles depends on a number of factors including vehicle movement (particularly traffic speed), street maintenance, season and associated meteorological parameters and speed of traffic (Etyemezian et al., 2003; Gertler et al., 2006; Thorpe et al., 2007; Bhaskar and Sharma, 2008; Kaunhaniemi et al., 2011; Laidlaw et al., 2012; Majumdar et al., 2012). Nicholson (1988) presents a detailed review of the mechanisms of re-suspension of road dust. Thorpe et al. (2007) reported a strong association between heavy duty traffic and re-suspension in the UK with wind speed not found to be a strong influence. In addition, precipitation was found to have no influence on the amount of re-suspension. In somewhat related research, street washing has been reported to be ineffective for PM control based on experiments in Spain (Karanasiou et al., 2012). However, in Sweden, road wetness was found to be an important factor in the amount of re-suspension (Omstedt et al., 2005), but this appears

Table 3
Ratio of Cu to Sb characteristic of brake wear particles in ambient PM reported in the literature.

Reference	Location	Cu:Sb ratio
Stechmann and Dannecker (1990)	Hamburg, Germany	5.6
Adachi and Tainosho (2004)	Kobe, Japan	1.3
Lin et al. (2005)	Taiwan	9.1 ± 1.79 (coarse fraction)
		2.17 ± 0.83 (fine fraction)
Wahlin et al. (2006)	Copenhagen, Denmark	4.4 ± 0.3
Hjortenkrans et al. (2007)	Stockholm, Sweden	5.3
Fabretti et al. (2009)	Nice, France	7.0 ± 1.3
Dongarra et al. (2009)	Palermo, Italy	4.9 (coarse fraction)
		2.5 (fine fraction)
Amato et al. (2011b)	Barcelona, Spain	8.0 ± 1.5 (roadside)
		6.8 ± 1.9 (urban)
Bukowiecki et al. (2010)	Zürich-Weststrasse, Switzerland	6.2 (coarse fraction)
Pey et al. (2010)	Barcelona, Spain	9 (coarse fraction)
		10 (fine fraction)
Gietl et al. (2010)	London, UK	9.1

related to road sanding and the use of studded tyres in Sweden. It is also important to note that the residence time of PM_{10} on paved roads (travel lanes) has been estimated as a few hours (Etyemezian et al., 2003). In the case of studded tyres, interaction between tyres and the road pavement can generate particles less than 0.1 μm (Gustafsson et al., 2008) and such tyres have been shown to increase PM_{10} emissions by a factor of 1.5 (Tervahattu et al., 2006). The type of pavement is another important factor for road dust emissions. Granite pavements have been reported to emit more PM compared to quartzite pavements (Kupiainen et al., 2005; Gustafsson et al., 2009). Tervahattu et al. (2006) also explained the higher PM_{10} concentration attributed to use of anti-skid aggregate using the *sandpaper effect*, where the aggregates used to prevent skidding generate PM_{10} particles and lead to further particle emissions from the pavement.

Pristane, phytane, hopanes, steranes in addition to unresolved complex mixture (UCM) were reported in road dust particles which were found to be enriched in biogenic component by Omar et al. (2007). High concentrations of phenanthrene, fluoranthene and pyrene along with high molecular weight (4–7 ring) PAHs were reported in road dust samples in Taiwan (Fang et al., 2004), China (Liu et al., 2007; Han et al., 2009) and Egypt (Hassanien and Abdel-Latif, 2008) respectively. The spatial distribution of PAHs was reported to be affected by wind direction Liu et al. (2007) and total organic carbon content by Liu et al. (2007) and Han et al. (2009). Mono- (α- and β-glucose) and disaccharides (sucrose and mycose) were used as markers for road dust re-suspension for source apportionment by Omar et al. (2007).

Han et al. (2007) analysed suspended dust in Beijing and reported concentrations of elements such as Ca, S, Cu, Zn, Ni, Pb, and Cd to be much higher than the crustal abundances. Cu, Zn, Ni, and Pb were attributed to traffic emissions together with coal burning, and ions such as Ca^{2+}, SO_4^{2-}, Cl^-, NO_3^-, K^+, and Na^+ were also found to be present in re-suspended dust. Luo et al. (2011) observed that the concentrations of trace metals contributed by vehicle exhaust and tyre abrasion, i.e. Pb, Zn and Cu have been reported to be much higher in roadside samples than the concentrations of these elements in background soils in China. Wei and Yang (2010) analysed urban soil, urban road dust and agricultural soil in Chinese cities and found relatively higher concentrations of Cd, Cu, Pb and Zn in urban soil samples and Cu, Cr, Pb, Zn, Ni and Cd in urban road dusts. The high concentrations of metals were attributed to anthropogenic sources − road traffic and industrial emissions. Higher concentration levels for several trace elements were reported for roadside dust compared to dust collected at a distance of 100 m in Bosnia, and particularly high concentrations were reported for Hg and Co and were attributed to vehicle exhaust and brake pads respectively. Elements such as platinum, palladium and rhodium have also been reported to be present in road dust and are attributed to the catalytic converters (Prichard and Fisher, 2012). Duong and Lee (2011) analyzed heavy metal contamination in road dust in high traffic areas in Korea and concluded that the concentrations of heavy metals are much higher in high traffic areas in relation to a background site. Their analysis also confirmed a high degree of correlation between the concentration of heavy metals at a specific location and the traffic throughput in the area. Martuzevicius et al. (2011) analysed street dust and reported that it can be a major source for particle-bound PAHs together with other PM. Han et al. (2009) reported a high correlation between PAHs and OC in road dust, and traffic emissions were identified as a major contributor to PAHs in road dust using diagnostic ratio analysis. Mathematical models have been proposed for estimation of PM emissions due to road dust although they do not include emissions due to wear of brake/tyre materials (Omstedt et al., 2005; Ketzel et al., 2007; Berger and Denby, 2011; Kaunhaniemi et al., 2011).

Aatmeeyata and Sharma (2009) reported a bimodal number and mass distribution (0.3 μm and 4–5 μm) for PM_{10} generated due to surface (concrete pavement) and tyre wear. Duong and Lee (2011) reported a multi-modal distribution for road dust particles (based on weight) collected from the roadside with a majority of the particles between 180 and 850 μm in Ulsan, Republic of Korea. Chen et al. (2006) reported a bimodal (5–10 μm and >30 μm) mass size distribution for road dust particles in Beijing (China).

3. Exhaust emissions

Motor vehicles are an important source for carbonaceous aerosols particularly for the particles in the fine size range (aerodynamic diameter < 2.5 μm) (El Haddad et al., 2009; Kam et al., 2012; Keuken et al., 2012). PM emissions from vehicles depend on the engine type and age and maintenance and contain carbon in the form of OC and EC with smaller amounts of trace metals and ions (Brook et al., 2007; Robert et al., 2007a; Fulper et al., 2010; Peltier et al., 2011). For example, Robert et al. (2007a,b) reported that that emission rates for ultrafine and fine particles for heavy duty diesel vehicles (HDDVs) are one order of magnitude higher than LDGVs. Similarly, Chen et al. (2013) reported 4-times higher diesel EFs compared to gasoline EFs in a tunnel environment. Higher emissions from the high-emitting (smoker) vehicles have previously been attributed to lubricating oil (Fujita et al., 2006). Emissions from diesel and gasoline vehicles are different in terms of composition as diesel engines emit both a greater mass of PM and a larger number of ultrafine particles (UFPs) compared to gasoline vehicles (Rose et al., 2006). Gasoline engines are known to release a higher fraction of OC while diesel engines emit more EC (Watson et al., 1994; Weingartner et al., 1997; Ntziachristos et al., 2007) although Sodeman et al. (2005) demonstrated a dominance of EC in the ultrafine range in gasoline vehicles. Zhu et al. (2010) reported $PM_{2.5}$ emissions from diesel and gasoline vehicles to be rich in different fraction of EC and OC; while EC2 and OC2 fraction were higher in case of diesel vehicles, EC3 and OC3 were higher in case of gasoline vehicle emissions. Vehicles are also a major source of *n*-alkanes, and diesel engines are known to emit more *n*-alkanes compared to gasoline engines (Rogge et al., 1993). Results from tunnel studies and chassis dynamometer analyses have shown that lower molecular weight PAHs such as phenanthrene and fluoranthene are characteristic of diesel-derived aerosols whereas gasoline engines contributed more to higher molecular weight PAHs such as benzo(ghi)perylene (Smith and Harrison, 1996). PAH emissions from gasoline engines are reported to be one order of magnitude higher than the diesel engine emissions integrated across the U.S. vehicle fleet (Lough et al., 2005). A detailed description of diesel particulate matter is presented in Maricq (2007). The situation in relation to exhaust emissions from traffic is constantly changing as abatement technologies develop. In particular the adoption of diesel particle filters is greatly reducing mass emissions.

Both trace elements and organic compounds are used as source markers for vehicle emissions. However, since the removal of lead from gasoline, trace metals have proved far less useful as a tracer of engine exhaust (Harrison et al., 2003). Organic compounds released from vehicles are particularly useful markers for conducting receptor modelling analyses since such markers help in distinguishing the vehicular emissions from other sources. The most commonly used molecular markers for vehicular emissions are hopanes and steranes and PAHs such as benzo(ghi)perylene which are attributed to lubricating oil (Rogge et al., 1993; Schauer et al., 1996; Cass, 1998; Lough et al., 2007) and due to their source, the distribution of hopane emissions from vehicles is suggested to be independent of fleet composition (He et al., 2008). It has also been

suggested that hopanes and steranes (derived from lubricating oil) and PAHs (reactions at high temperature and fuel combustion) are indicative of different processes during the organic carbon formation in vehicle engines (Lee et al., 1995; Fujita et al., 2006; Riddle et al., 2007; Hanedar et al., 2008).

A number of PAHs have been reported to be present in vehicle exhaust including benzo(e)pyrene, benzo(a)pyrene, indeno(1,2,3-cd) pyrene, coronene and benzo(ghi)perylene (Rogge et al., 1993; Lough et al., 2007; Riddle et al., 2007). However, Gao et al. (2011) attributed indeno(1,2,3-cd)pyrene and benzo(ghi)perylene to biomass combustion in Guangzhou (China). PAHs are found predominantly in the smallest size fraction (<0.4 μm), and most of the high molecular weight PAHs are found in the fine particle fraction (Hien et al., 2007). Diagnostic ratio analysis including the ratio between concentrations of indeno(1,2,3-cd) pyrene to the sum of indeno(1,2,3-cd) pyrene and benzo(ghi)perylene is also used for distinguishing between gasoline and diesel emissions (Chellam et al., 2005; He et al., 2006; Shen et al., 2010; Ancelet et al., 2011; Kim et al., 2012). Ratios between methylphenanthrenes/phenanthrene has also been used for estimation of the contribution of traffic to PAH concentrations in ambient air (Lim et al., 1999). However, it is important to remember that PAHs can be altered by atmospheric chemical reactions (Wu et al., 2007). Also, emission profiles for PAHs are difficult to generalize since they vary across the different vehicle classes (Miguel et al., 1998). Congener profiles of PAH from road traffic have been derived from tunnel studies (e.g. Smith and Harrison, 1996) and from roadside concentrations from which the local background is subtracted (Mari et al., 2010). Nielsen (1996) also prepared a traffic PAH profile for Copenhagen (Denmark) using a roadside study.

Elemental markers which have been used for vehicular emissions include Cu, Mn, Fe, Zn, Ba, Sn, Ni, Mo and Sb (Lough et al., 2005; Almeida et al., 2006; Birmili et al., 2006; Crawford et al., 2007; Dongarra et al., 2009; Fabretti et al., 2009; Gietl et al., 2010; Amato et al., 2011a,b). Metals can be emitted from various exhaust-related sources including fuel and lubricant combustion, catalytic converters, particulate filters and engine corrosion (Lough et al., 2005; Pulles et al., 2012; Sysalova et al., 2012; Varrica et al., 2012) but many of these appear most likely to arise from non-exhaust sources. Ni and V have also been reported to be present in emissions due to oil combustion (Pey et al., 2010). Cheng et al. (2010) reported higher emission rates for elements such as Br, Ba, Sb and V for gasoline and liquefied petroleum gas (LPG) engines compared to diesel engines in Hong Kong and indicated potential use of these species as markers for gasoline/LPG engines. Ba, Cd, Zn, Sb and V in nanoparticles have been reported to be strongly associated with diesel fuel whereas Cu, Mn and Sr in the particles <0.1 μm have been found to be associated with gasoline (Lin et al., 2005). Metal concentrations from diesel and gasoline vehicles are reported to vary over two orders of magnitude (Pulles et al., 2012). In a traffic emissions study in New York (USA), Ba, Br and black carbon (BC) were used as tracers for gasoline/diesel combustion (Peltier et al., 2011). However, the levels of trace elements emitted in the exhaust are very low, and great care is needed to distinguish them from non-exhaust traffic emissions and other sources. In addition, different authors have attributed metals to different sources which renders precise source attribution with trace metals alone very difficult unless detailed local information on source particle composition is available (e.g. Pant and Harrison, 2012).

4. Particle size distributions

The size distribution of a particular element or compound not only influences the potential health impact (in terms of respiratory

deposition – Harrison et al., 2010), but also influences the extent of atmospheric dispersion (Allen et al., 2001). Size distributions can be measured in terms of mass, number or surface area (Harrison et al., 2000). A number of factors can influence the size distribution including, but not limited to vehicle fleet mix, road type and grade, meteorological conditions (e.g. season type) and distance from road (Zhu et al., 2006; Beddows et al., 2009; Hays et al., 2011; Song and Gao, 2011). For example, differences were observed in the modal structure of particles close to the freeway and at a crossroad in Prague and the difference was attributed to the higher concentration of UFPs in the freeway due to the high volume of traffic (Ondráček et al., 2011). However, in a study in the USA, Padro-Martinez et al. (2012) concluded that distance from the highway does not impact the particle size distribution. Many other studies (e.g. Shi et al., 1999) have reported the opposite. The behaviour of nanoparticles after emission is discussed later in this section. Size distributions can be useful in ascertaining the sources of exhaust and non-exhaust particle emissions. For example, sampling on a busy London (UK) highway, Harrison et al. (2011) used Positive Matrix Factorization (PMF) to disaggregate the particle size distribution which demonstrated a nucleation mode centered on around 20 nm and a solid particle mode at 50–60 nm in the number distribution. However, in using such data, it should be borne in mind that the size distributions measured by cascade impactors may be modified from those in ambient air as an artefact caused by the semi-volatile nature of the PAHs, hopanes and steranes.

4.1. Mass size distributions

Based on size distribution analysis for trace metals found in atmospheric aerosols at urban and background sites in the UK, it was concluded that there are three key categories; metals with most of the mass in the accumulation (fine) mode (e.g. Cs, Sn, Pb) with an additional minor mode, metals with mass distributed between fine, coarse and intermediate mode (e.g. Ni, Zn, Cu, Mn) and metals with most of the mass in the coarse range (Fe, Ba, Sr) (Allen et al., 2001; Birmili et al., 2006). Song and Gao (2011) reported similar results for size-segregated PM at a highway site in the USA with a bimodal mass-based distribution with the two peaks at 0.32–0.56 μm and 3.2–5.6 μm Table 4 summarizes some of the published data for traffic-related elements.

Similar observations regarding mass-based size distributions for trace metals were reported by Samara and Voutsa (2005) for a traffic site in Greece. Metals such as Ba, Pb and Zn were found to exhibit bimodal distributions whereas some others such as Cr and V showed trimodal distributions in Southern Taiwan (Lin et al., 2005). The presence of an Aitken mode in the ultrafine range may be indicative of a contribution from vehicular emissions, both diesel and gasoline. Hays et al. (2011) reported erratic mass size distributions for various elements including Ni, Cu, Cd, and Zn and attributed this to a range of emission sources near highways in North Carolina (USA). A peak was observed at 80 nm for soot particles in a street canyon corresponding to diesel engine emissions in Leipzig, Germany and the concentration of soot particles was reported to be influenced by traffic volume (Rose et al., 2006).

4.2. Number-size distributions

Ultrafine particles are characterized as particles with diameter less than 100 nm diameter and are typically measured as particle number concentration (PNC) (Charron and Harrison, 2009). Such particles may have a short lifetime and as a result, the spatial variability of the particles differs based on proximity to the emission source (Puustinen et al., 2007; Cyrys et al., 2008; Charron and Harrison, 2009). Traffic has been reported as one of the major

Table 4
(a) Results from mass size distribution analysis for traffic-related PM as reported in the literature (in μm). (b) Results from mass size distribution analysis for traffic-related elements as reported in the literature (in μm).

(a)

Reference	Location	Site type	Component	Distribution type	Primary peak
Allen et al. (2001)	USA	Tunnel	PM_{10}	Unimodal	0.1–0.18
Hildemann et al. (1991)	USA	Freeway	PM	Trimodal	Ultrafine (<0.2), fine (0.2–1) and coarse (1–10)
Robert et al. (2007a,b)	USA	Dynamometer	$PM_{1.8}$ (HDGV)	Unimodal	0.1–0.18
			$PM_{1.8}$ (LDGV)	Unimodal	0.1–0.18

(b)

Reference	Location	Metal(s)	Distribution type	Primary peak	Secondary peak
Stechmann and Dannecker (1990)	Germany	Cu, Ba	Unimodal	2.5	
Funasaka et al. (1998)	Japan (tunnel)	EC	Unimodal		
		OC	Bimodal	1	2–10
Lin et al., 2005	Taiwan	Si, Al, Fe, Ca	Bimodal	3.2–5.6	Nano size range
		Ba	Bimodal	1.0–1.8	0.18–0.32
		Pb	Bimodal	3.2–5.6	0.32–0.56
		Zn	Bimodal	1.0–1.8	0.32–0.56
		Mn, Cu, Cd, Sr	Bimodal		
		Cr, V	Trimodal		
Lough et al. (2005)	USA	Cu	Bimodal	3.0	0.1
Wahlin et al. (2006)	Denmark	Ba, Sb	Unimodal	3	
		Cu	Bimodal	3	0.1
Hien et al. (2007)	Vietnam	PAHs in $PM_{2.5}$	Bimodal	Coarse and fine	
Riddle et al. (2007)	USA	Hopanes and steranes in $PM_{1.8}$	Unimodal	0.10–0.18	0.32 (vehicles without catalysts
Wang et al. (2009)	China	n-Alkanes, PAHs and Hopanes	Unimodal	0.7–1.1	
Gietl et al. (2010)	UK	Ba, Fe	Unimodal	1.2–7.2	
		Cu, Sb	Bimodal	1.2–7.2	0.2–0.4
Huang et al. (2012)	China	n-Alkanes	Unimodal	0.32–0.56	
		PAHs	Unimodal	0.56–1.0	
		Hopanes	Unimodal	0.56–1.0	3.2 (in some cases)
Ondráček et al. (2011)	Czech Republic (freeway)	Fe, Cu, Mn, Zn	Unimodal	2.5	
		Si, Al, Ca and coarse mode K	Unimodal	5	
Song and Gao (2011)	USA	Al, Fe, Cu, Sb, Sc, Mn	Bimodal	3.2–5.6	1.0–1.8
		Cd	Bimodal	0.18–0.56	3.2–5.6
		Pb, Zn	Trimodal	0.32–0.56	1.0–1.8, 3.2–5.6
		Ni, V	Bimodal	0.18–0.32	3.2–5.6

sources of UFPs (Abu-Allaban et al., 2002; Nanzetta and Holmen, 2004; Westerdahl et al., 2009) and particle number is linked to fresh vehicle exhaust emissions (Rodriguez et al., 2007). Number size distributions are heavily influenced by proximity to traffic (Rose et al., 2006; Kerminen et al., 2007; Beddows et al., 2009; Keuken et al., 2012). As a consequence of this, concentrations of organic compounds have also been reported to be highly variable for near-highway sites (Sun et al., 2012. Highest PNC has been reported during morning rush hours in Germany (Wehner et al., 2002) and Spain (Perez et al., 2010). A significant difference has also been reported between weekdays and weekends, highlighting the influence of traffic (Morawska et al., 2002; Wehner et al., 2002; Van Poppel et al., 2012). A significant difference was observed in the particle number size distribution in the Kingsway tunnel in the UK (peak at 30–40 nm) and Plabutsch tunnel in Austria (80–100 nm) and this was attributed to the differences in traffic intensity and the fleet composition, i.e. proportion of HDVs and LDVs (Imhof et al., 2005a). It is important to note that although PNC is heavily influenced by road traffic, it is also influenced by other sources including atmospheric nucleation processes (Imhof et al., 2005a; Reche et al., 2011).

The typical particle number distribution of a vehicle has 3 modes: nucleation mode (particles < 50 nm), consisting of particles formed due to condensation of exhaust particles; accumulation mode (50–300 nm with a peak at ∼80–100 nm) consisting of soot particles, and a coarse mode (1 μm–10 μm) consisting of particles

generated due to abrasion and wear and tear of brake pads and tyres and re-suspension of road dust (Kittleson et al., 2004; Imhof et al., 2005a). The number size distribution for diesel engine emissions has been reported as lognormal with mean diameter between 60 and 120 nm whereas gasoline engine emissions have asymmetric distributions with a mean diameter between 40 and 80 nm (Harris and Maricq, 2001). PNC shows higher variability compared to particle mass (Weijers et al., 2004). While nuclei mode particles only constitute about 10% of the total particle mass, they contribute more than 90% to PNC (Kittleson et al., 2004). Fushimi et al. (2008) reported the presence of a nuclei mode at 20 nm in the number distribution for PM at roadside while no such mode was observed at the background site in Kawasaki, Japan. PNC was reported to be significantly higher in a street canyon compared to an urban background location in a study in Leipzig, Germany (Wehner et al., 2002).

The particle number distribution for vehicular exhaust emissions measured in ambient air is typically bimodal with one peak at 10–40 nm and another one at 50–90 nm (Charron and Harrison, 2009). Gasoline exhaust particles fall in the size range of 20–60 nm whereas diesel exhaust particles are in the size range of 20–130 nm (accumulation mode between 50 and 90 nm and a smaller nucleation mode) (Zhu et al., 2002b; Rissler et al., 2012). Such bimodal distributions have been reported for vehicular emissions in many parts of the world including Korea (30–50 nm

and 100–400 nm for diesel engines) (Lim et al., 2008), Czech Republic (25–30 and 60–70 nm) (Ondráček et al., 2011) and United Kingdom (13.3 nm and 86.6 nm) (Kumar et al., 2008). On the other hand, multi-modal distributions have been reported in several cities and the larger number of modes has been attributed to mixing of traffic and background aerosol (Ondráček et al., 2011). Lingard et al. (2006) reported a four-mode particle number size distribution for an urban roadside site in Leeds (UK) with particles in modes II and III to be highest during the rush hours indicating a traffic source. The four modes were classified as sub-11 nm nucleation mode, super-11 nm nucleation mode, Aitken mode and accumulation mode and were attributed to photolysis, diesel engine emissions and secondary aerosols (Lingard et al., 2006). Rosenbohm et al. (2006) observed a bimodal (20 nm and 80 nm) size distribution at a site downwind of the motorway whereas a unimodal distribution (80 nm) was observed for an upwind site. El Haddad et al. (2009) reported a single mode (50–90 nm) distribution for PM in a road tunnel in Marseille, France and Abu-Allaban et al. (2002) observed the highest PNC in the nucleation mode (11–17 nm) in a roadway tunnel in USA. A bimodal (11 nm and 80 nm) distribution was reported for $PM_{0.5}$ emissions during a vehicle chase experiment (Wang et al., 2011). Although the magnitude of PNC emissions from a gasoline spark ignition engine exhaust is smaller than from a diesel engine, emissions from gasoline engines are comparable to diesel engines in the cases of acceleration and freeway cruising (Kittleson et al., 2004). Klems et al. (2010) also reported higher PNC at the time of acceleration at traffic lights.

Significant differences have been reported between dynamometer and real-world studies on PNC. A mean diameter of 15 nm for PM emissions was measured in Leipzig, Germany in a street canyon though some lab-studies on gasoline and diesel engines have shown mean diameters between 40 and 70 nm (Wehner et al., 2002). It is important for dynamometer studies to reflect adequately the dilution properties of the atmosphere or they may suppress the nucleation mode which is formed during the initial dilution of the exhaust plume (Shi and Harrison, 1999).

For traffic emissions, distance from the road is also reported to cause changes in PNC due to processes other than dilution (Zhu et al., 2006). This feature is of considerable interest; particularly from an epidemiological perspective as this would mean that exposure from near-roadway sites can be much different from other sites. HEI (2010) reported the key exposure zone for traffic emissions to be within 300–500 m of a roadway. In Los Angeles, both PNC and size distribution changed with an increase in distance from the road, particularly for particles with a diameter less than 30 nm and it has been suggested that this drastic change in PNC is caused by coagulation and atmospheric dilution (Zhu et al., 2002a). Padro-Martinez et al. (2012) found PNC to be at least two times higher at 0–50 m from the highway compared to further distances while Zhu et al. (2002b) observed the PNC to be 25 times higher near the freeway compared to background locations. Rose et al. (2006) also observed a decrease in soot particle number concentration with increasing distance from the street canyon which was attributed to coagulation processes. Kittleson et al. (2004) observed a decline in the nuclei mode particles as the distance from the highway increased and recently, Canagaratna et al. (2010) also reported a decrease in PNC with increasing distance away from the roadway. Dilution processes will inevitably cause a reduction in PNC with distance from the road, as they do for other traffic-generated pollutants (e.g. NO_x, BC, etc.). Coagulation causes an increase in particle size, and a decrease in number, but is unlikely to be a significant influence close to the road due to an insufficient number concentration to drive the process, and short travel times. An observed increase in particle diameters with distance from a road is most probably the result of mixing of traffic emissions with a coarser background aerosol leading to a large

modal diameter (Shi et al., 1999). The nucleation mode traffic particles are semi-volatile and this can lead to shifts in size distribution (typically to smaller sizes) and particle loss due to evaporation (Dall'Osto et al., 2011; Harrison et al., 2012a).

Charron and Harrison (2003) reported that increasing wind speeds and lower temperatures can increase the abundance of smaller nucleation mode particles (diameter of sub-30 nm) and this could be due to the greater dilution and lower condensation sink. Ruellan and Cachier (2001) also reported an influence of wind speed and boundary layer height on PNC. Kumar et al. (2008) reported traffic volume and wind speed as the key variables influencing PNC in street canyon conditions and they observed a linear dependence of PNC on traffic volume with a higher correlation in the case of smaller particles (diameter < 300 nm) and a weaker correlation in the case of larger particles for which other sources are large relative contributors. Highest PNC was observed for calm winds and lowest for wind speeds above 1.6 m s^{-1} in Somerville (USA) (Padro-Martinez et al., 2012). PNC was found to be correlated with traffic intensity in France (Ruellan and Cachier, 2001; El Haddad et al., 2009; Perez et al., 2010) and in Los Angeles, PNC was reported to be influenced by traffic density and wind speed and direction (Zhu et al., 2002b). In a study in London, Milan and Barcelona, Rodriguez et al. (2007) concluded that vehicular exhaust emissions largely influence ultrafine particles and found a unimodal distribution for particle number with a mode at 37 nm. Kittleson et al. (2004) observed a direct influence of increasing vehicle speed on nuclei mode particles. Decreasing the engine load from 100% to about 60% was reported to decrease the PNC as well as EC and OC emissions from diesel vehicles (Lim et al., 2008). Morawska et al. (2002), on the other hand, indicated a 70% increase in PNC with a 50% increase in traffic flow. While roadside PNC measurements have indicated an influence of temperature on the concentration, similar studies in laboratory conditions using chassis dynamometers have not shown such influence (Abu-Allaban et al., 2002; Kittleson et al., 2004). Abu-Allaban et al. (2002) reported an inverse relationship between ambient air temperature and the diameter of nucleation mode particles. Using modelling analysis, Jayaratne et al. (2009) observed an increase in PNC at a signalized pedestrian crossing due to the stop and go nature of traffic at the signal and the increase was found to be higher for HDVs compared to LDVs. This is similar to the observations from Abu-Allaban et al. (2002) and Wang et al. (2010) who observed higher concentrations arising from HDVs compared to LDVs in USA and Denmark respectively. Rosenbohm et al. (2006) reported a correlation between PNC and NO in Heidelberg (Germany) although no correlation was observed for PNC and PM_{10} concentration. Similarly, Padro-Martinez et al. (2012) also reported a similar spatial trend for PNC, NO_x, particulate PAHs and BC in the USA while a different trend was reported for $PM_{2.5}$.

Thus, key factors that influence the particle number concentration measurements include wind speed, traffic density and the distance from source at which the measurements are made. Nanzetta and Holmen (2004) observed differences in the PNC across different seasons and concluded that results for one season cannot be used for generalization of PNC.

5. Methodologies for assessment of traffic emissions

There are several ways of evaluating emissions from road vehicles including tests with dynamometers, measurements in tunnels/highways and near roadside measurements and road simulator tests (Fig. 1) (McLaren et al., 1996; Phuleria et al., 2007; Handler et al., 2008; Gustafsson et al., 2008; El Haddad et al., 2009; Yan et al., 2009; Sjodin et al., 2010). Table 5 presents a qualitative description of measurements using dynamometers under lab
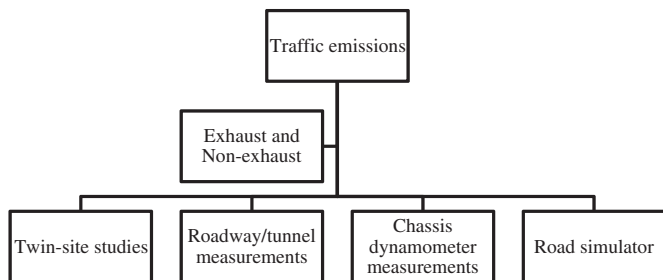
Fig. 1. Methods for direct road traffic emissions analyses.

conditions and tunnel/roadway/twin-site studies under ambient conditions. In recent years, significant differences have been observed between laboratory-tested and real-world mixed source traffic emissions (Gertler et al., 2002; Yan et al., 2009; Ancelet et al., 2011). Near roadside/kerbside, traffic tunnel and highway measurements are reported to be more realistic since they represent mixed fleet emissions under real-world driving conditions (Phuleria et al., 2007). Beddows and Harrison (2008) compared particle number EFs measured in dynamometer studies with those estimated in roadside studies, finding generally good agreement overall. However, Canagaratna et al. (2010) reported the EFs calculated using on-road measurements to be nearly 50% lower than EFs measured using a dynamometer.

An emission factor is defined as "the mass (or number) of pollutant released per unit time/distance travelled or mass of fuel used" and typically increases with vehicle speed for LDVs as well as HDVs (Jones and Harrison, 2006). Characterization of emissions from road vehicles includes analysis of size distributions and chemical composition (organic, inorganic and trace metal) and can lead to generation of EFs. EFs typically depend on conditions at the sampling site, traffic intensity and modal shares of HDVs and LDVs (Staehelin et al., 1995; Imhof et al., 2005a). A recent review article on development of emission factors delves into this issue in greater detail (Franco et al., 2013).

EFs can be measured using a number of different methods including dynamometer studies (Oanh et al., 2010; Chiang et al., 2012), roadway tunnel studies (Gertler et al., 2002; Handler et al., 2008; Mancilla and Mendoza, 2012), on-road chase experiments (Wang et al., 2011) and remote sensing (Zhang et al., 1995). Phuleria et al. (2007) reported overall agreement between EFs determined at tunnel and roadway sites in USA.

Two of the most common methods for EF expression are fuel-based EF (Ban-Weiss et al., 2010; Wang et al., 2011) and vehicle kilometre travelled (VKT)-based calculation (Abu-Allaban et al., 2002; Kittleson et al., 2004; Jones and Harrison, 2006; Amato et al., 2012). Phuleria et al. (2006, 2007) and Bukowiecki et al. (2010) describe the methodology of EF derivation in greater

detail. Fuel-based EF can be defined as "grams of pollutant emitted for every kilogram of fuel burned" and can be calculated for various pollutants using CO_2 and CO as a measure of fuel burned (Liacos et al., 2012). The concentration of pollutant with respect to the concentration of $CO_2 + CO$ can be used to calculate the fuel burnt using the following formula (Miguel et al., 1998; Liacos et al., 2012):

$$EF_P = 106 \times \left(\frac{\Delta P}{\Delta CO_2 + \Delta CO} \right) w_c \tag{1}$$

where EF_P is the emission factor for the pollutant ($\mu g \ kg^{-1}$ fuel burned), ΔP ($ng \ m^{-3}$), ΔCO_2 and ΔCO ($\mu gC \ m^{-3}$) are the increments over background concentration and w_c is the fraction of carbon weight in the fuel.

One of the other most commonly used methods is the derivation of PM EFs using NO_x EFs, which are calculated as described in the following equation (Gehrig et al., 2004; Imhof et al., 2005b). It is assumed in this case that both NO_x and PM have similar dilution.

$$D = \frac{NLDV \times EF \ NO_x LDV + NHDV \times EF \ NO_x \ HDV}{\Delta NO_x} \tag{2}$$

where D is the dilution rate ($m^2 \ h^{-1}$), n is the number of vehicles (h^{-1}), and $EF \ NO_x$ is the emission factor for NO_x in $mg \ km^{-1}$.

Using the dilution rate, the EF (per hour) can be calculated per vehicle based on:

$$EF_x = \frac{\Delta Concentration_x \times D}{n total} \tag{3}$$

where $\Delta Concentration_x$ is the difference in concentration of species $\times$ between roadside and background and n is the total number of vehicles crossing the sampling point per hour. The respective contribution of HDVs and LDVs can then be estimated with multiple linear regression (MLR) (Gertler and Pierson, 1996; Imhof et al., 2005b). Table 6 presents some EFs reported for PM_{10}, $PM_{2.5}$ and particle number concentration in the literature.

5.1. Tunnel/roadway measurements

EF calculations from tunnel studies are typically based on calculation of the difference in concentration between entrance and exit (Weingartner et al., 1997; Abu-Allaban et al., 2002), and Gertler and Pierson (1996) describe the methodology for calculation of emission factors using tunnel measurements. Chang et al. (1981) developed a model relating air quality in the tunnel to vehicular emissions which can also be used for calculation of emission factors using tunnel measurements. For the tunnel studies, "the velocity and concentration are assumed to be the same across the tunnel's cross-section and the difference between the

Table 5
Comparison between dynamometer and tunnel/roadway measurements (Allen et al., 2001; Imhof et al., 2005a; Phuleria et al., 2006, 2007; Handler et al., 2008; He et al., 2008; El Haddad et al., 2009; Sanchez-Ccoyllo et al., 2009; Franco et al., 2013).

	Dynamometer measurements	Tunnel/roadway measurements
Testing conditions	Precise and controlled; different test cycles and driving conditions can be analysed; can include cold-start emissions	Ambient conditions cannot be physically controlled; cannot include cold-start emissions. Better known boundary conditions and dilution effects. In some cases, atmospheric conditions in tunnel may not be the same as ambient environment
Representativeness for the vehicle fleet	Low, since the tests are conducted on a sub-set of the in-use fleet and variations in engine type, vehicle age and maintenance and mixing of emissions from different vehicles etc are not accounted for. Such studies do not account for particle aging effects	High, since the measurements are made in ambient environment and for in-use mixed fleet
Emission type accounted for	Exhaust	Exhaust and non-exhaust

Table 6
Emission factors as reported in the literature (selected studies).

Particle number

Reference	City, country	Study type	Particle number EF
Gertler et al. (2002)	USA	Tunnel (mixed LDV and HDV)	5×10^{12} particles veh^{-1} km^{-1} (low HDV)
			$2–3 \times 10^{13}$ particles veh^{-1} km^{-1} (high HDV)
Abu-Allaban et al. (2002)	USA	Tunnel	$5.2–5.4 \times 10^{13}$ particles veh^{-1} km^{-1} (low HD fraction)
			$2.1–3.1 \times 10^{14}$ particles veh^{-1} km^{-1} (high HD fraction)
Jamriska et al. (2004)	Australia	Tunnel (diesel powered buses)	$3.11 \pm 2.41 \times 10^{14}$ particles km^{-1}
Kittleson et al. (2004)	USA	Roadside (particles > 3 nm)	$1.9–9.9 \times 10^{14}$ particles km^{-1} (gasoline-dominant fleet)
Jones and Harrison (2006)	UK	Roadside	5.84×10^{13} particles veh^{-1} km^{-1} (LDV)
			6.36×10^{14} particles veh^{-1} km^{-1} (HDV)
Westerdahl et al. (2009)	China	Roadside – LDV (prevalent during daytime)	1.8×10^{15} particles kg^{-1} fuel
		Roadside – HDV (prevalent during nighttime)	1.1×10^{16} particles kg^{-1} fuel
Wang et al. (2009)	Denmark	Highway	$(215 \pm 15) \times 10^{12}$ particles veh^{-1} km^{-1}
Ban-Weiss et al. (2010)	USA	Tunnel-LDV	$(3.9 \pm 1.4) \times 10^{14}$ particles kg^{-1} fuel
		Tunnel-HDV	$(3.3 \pm 1.3) \times 10^{15}$ particles kg^{-1} fuel
Wang et al. (2011)	China	Vehicle chase	7.1×10^{16} particles kg^{-1} fuel

Particle mass

Reference	City, country	Study type	PM size range	EF
Weingartner et al. (1997)	Switzerland	Tunnel (mixed LDV and HDV with more LDV)	PM$_3$	310 mg km^{-1} (63% vehicles running on diesel)
Gillies et al. (2001)	USA	Tunnel (mixed LDV and HDV)	PM$_{2.5}$	0.052 ± 0.027 g km^{-1}
			PM$_{10}$	0.069 ± 0.030 g km^{-1}
Gertler et al. (2002)	USA	Tunnel (mixed LDV and HDV)	PM$_{2.5}$	0.062 ± 0.042 g km^{-1}
			PM$_{10}$	0.087 ± 0.054 g km^{-1}
Jamriska et al. (2004)	Australia	Tunnel (diesel powered buses)	PM$_{2.5}$	583 ± 451 mg km^{-1}
Grieshop et al. (2006)	USA	Tunnel	PM$_{2.5}$ (rush hour)	189 ± 23 mg kg^{-1} fuel
			PM$_{2.5}$ (mid-day)	158 ± 29 mg kg^{-1} fuel
			PM$_{2.5}$ (early morning-HDV)	437 ± 76 mg kg^{-1} fuel
Handler et al. (2008)	Austria	Tunnel	TSP	129 ± 45 mg veh^{-1} km^{-1}
			PM$_{10}$	62 ± 18 mg veh^{-1} km^{-1}
			PM$_{2.5}$	26 ± 10 mg veh^{-1} km^{-1}
Sanchez-Ccoyllo et al. (2009)	Brazil		Coarse	127 mg km^{-1}
			Fine	92 mg km^{-1}
Wang et al. (2010)	Denmark	Highway	PM$_{2.5}$	29 mg veh^{-1} km^{-1}
Chiang and Huang (2009)	Taiwan	Freeway tunnel	PM$_{2.5-10}$	18 ± 6.5 mg veh^{-1} km^{-1}
			PM$_{2.5}$	39 ± 11 mg veh^{-1} km^{-1}
Cheng et al. (2010)	Hong Kong	Tunnel, roadside sites and urban site	PM$_{2.5}$	257 ± 31 mg veh^{-1} km^{-1}
Bukowiecki et al. (2010)	Switzerland	Roadside	PM$_{10}$- LDV	24 ± 8 mg veh^{-1} km^{-1}
			PM$_{10}$ HDV	498 ± 86 mg veh^{-1} km^{-1}
Wang et al. (2011)	China	Vehicle chase	PM$_{0.5}$	2.35 g kg^{-1} fuel
Mancilla and Mendoza (2012)	Mexico	Tunnel	PM$_{10}$	22.8 ± 7.4 mg veh^{-1} km^{-1}
Chen et al. (2013)	China	Tunnel (mixed with higher proportion of gasoline vehicles)		687 mg veh^{-1} km^{-1} (summer)
				714 mg veh^{-1} km^{-1} (winter)

LDV and HDV vehicles (mass)

Reference	Country	Study type	PM size range	LDV EF	HDV EF
Weingartner et al. (1997)	Switzerland	Tunnel	PM$_3$	8.53 ± 0.47 mg km^{-1}	383.5 ± 10.7 mg km^{-1}
Gertler et al. (2002)	USA	Tunnel (mixed LDV and HDV)	PM$_{2.5}$	0.014 ± 0.013 g km^{-1}	0.135 ± 0.018 g km^{-1}
			PM$_{10}$	0.010 ± 0.011 g km^{-1}	0.181 ± 0.013 g km^{-1}
Abu-Allaban et al. (2003)	USA	Roadside (LD-Spark ignition and HD-diesel)	PM10	10–70 mg km^{-1} vehicle	60–570 mg km^{-1}
			PM$_{2.5}$	10–50 mg km^{-1} vehicle	60–480 mg km^{-1}
Chiang et al. (2012)	Taiwan	Dynamometer	PM$_{2.5}$	0.172 g km^{-1} (diesel)	
Oanh et al. (2010)	Bangkok	Dynamometer	PM$_{2.5}$	230 mg km^{-1}	176 mg km^{-1}
Jones and Harrison (2006)	UK	Roadside	PM$_{10}$	0.033 ± 0.006 g veh^{-1} km^{-1}	0.370 ± 0.032 g veh^{-1} km^{-1}
			PM$_{2.5}$	0.010 ± 0.004 g veh^{-1} km^{-1}	0.179 ± 0.022 g veh^{-1} km^{-1}
Liacos et al. (2012)	USA	Freeway	PM$_{2.5}$	151189.3 mg km^{-1} h (low HDV)	169629.5 mg km^{-1} h (high HDV)

Non-exhaust emissions

Reference	City, country	Study type	PM size range	Road dust	Brake wear	Tyre wear
Abu-Allaban et al. (2003)	USA	Roadside (LD-Spark ignition and HD-diesel)	PM$_{10}$ LDV	40–780 mg km^{-1} vehicle	0–80 mg km^{-1} vehicle	–
			PM$_{10}$ HDV	230–7800 mg km^{-1} vehicle	0–610 mg km^{-1} vehicle	
			PM$_{2.5}$ LDV	2–25 mg km^{-1} vehicle	0–5 mg km^{-1} vehicle	–
			PM$_{2.5}$ HDV	15–300 mg km^{-1} vehicle	0–15 mg km^{-1} vehicle	
Bukowiecki et al. (2010)	Switzerland	Roadside	PM10	27 mg veh^{-1} km^{-1}	15 mg veh^{-1} km^{-1}	
Amato et al. (2012)	Spain	Freeway	PM$_{10}$	22.7 ± 14.2 mg veh^{-1} km^{-1} travelled	–	–

influx and outflux is assumed to be the sum of emission rates of vehicles in the tunnel" (Rogak et al., 1998). The following formula is typically used for calculation of the EF (Gertler and Pierson, 1996; He et al., 2008):

$$EF = \frac{C_{out}V_{out} - C_{in}V_{in}}{NL} \tag{4}$$

where C is the concentration of the pollutant, V is the volume of air calculated using duration of sampling, cross-sectional area of tunnel and wind speed; N is the number of vehicles and; L is the distance between sampling locations at inlet and outlet of the tunnel.

Several studies have been conducted for measurement of PM emissions in roadway tunnels in different parts of the world including the USA (Fraser et al., 1998; Rogak et al., 1998; Gillies et al., 2001; Abu-Allaban et al., 2002; Gertler et al., 2002), China (He et al., 2006, 2008; Chiang and Huang, 2009), Europe (Stechmann and Dannecker, 1990; Smith and Harrison, 1996; Weingartner et al., 1997; Imhof et al., 2005a; Handler et al., 2008; El Haddad et al., 2009; Oliveira et al., 2011), New Zealand (Ancelet et al., 2011), Japan (Funasaka et al., 1998), Taiwan (Chiang and Huang, 2009), Chile (Caceres et al., 1998) and Mexico (Mancilla and Mendoza, 2012).

Although tunnel studies provide an opportunity to conduct measurements on a mixed vehicle fleet, it is important to remember that factors such as variations in speed, aerodynamic conditions in the tunnel and the fleet characteristics (i.e. proportion of HDVs and LDVs) can cause variability in measurements (Rogak et al., 1998; He et al., 2008). In addition, vehicles in the tunnel are often driving at a steady speed which does not happen under other road conditions where traffic follows a stop-and-go pattern (Gertler et al., 2002). Chirico et al. (2011) have shown that the distribution of organic compounds between particles and vapour is heavily affected by the high PM concentrations in a road tunnel, and thus may influence the estimate of emission factors for semi-volatile components.

PM_3 (particles with diameter < 3 μm) was analysed in the Gubrist Tunnel in Zurich, Switzerland and it was concluded that most of the particles are emitted from the tailpipe with a small contribution from tyre wear and road dust (Weingartner et al., 1997). While BC accounted for 31% of the total emissions, particulate PAHs accounted for 0.86% of PM_3 (Weingartner et al., 1997). In New Zealand, PAHs were reported to contribute nearly 0.10% of the total PM mass with a higher proportion of low molecular weight PAHs indicating a higher contribution of diesel emissions (Ancelet et al., 2011). In roadway tunnel studies in China, EC was found to contribute 63% and 45% to PM mass whereas OC contributed 34% and 31% (He et al., 2006, 2008). Ancelet et al. (2011), however, found OC to contribute between 32 and 46% to PM mass. Gillies et al. (2001) and He et al. (2008) have attributed higher $PM_{2.5}$ emissions to HDVs compared to light duty vehicles (LDVs) in USA and China respectively. Caceres et al. (1998) prepared a PAH source signature for traffic using tunnel measurements in Santiago de Chile whereas Smith and Harrison (1996) prepared a source profile fingerprint using tunnel measurements for traffic in Birmingham, UK. Funasaka et al. (1998) analysed emissions in a traffic tunnel in Osaka, Japan and EC and NO_x (NO as the main component) were found to be correlated with diesel traffic volume whereas OC did not show good correlation. In addition, the concentration of EC and SPM was observed to increase linearly with an increase in traffic volume. In Lisbon, (Portugal), nearly 84% of the total PAH mass in a road tunnel was reported to be less than 0.49 μm diameter by Oliveira et al. (2011).

Concentrations of trace elements were not found to be correlated with EC concentration in Zhujiang Tunnel (China) and the

authors concluded non-exhaust sources to be contributing to the trace element mass (He et al., 2008). A similar conclusion was reported by Funasaka et al. (1998) and Ancelet et al. (2011) who attributed coarse particles in a road tunnel in Osaka, Japan and inorganic elements such as Fe, Zn etc. respectively to non-exhaust emissions, primarily re-suspension of road dust. It should be noted, however, that the contributors to road dust in a tunnel will differ from those in a normal roadway environment and that both chemical composition and emission factors will be different. Partitioning of the semi-volatile components between particles and vapour can also be influenced by the high concentrations present (Chirico et al., 2011).

5.2. Twin-site studies

A number of papers have reported the estimation of the contribution of traffic emissions to total PM using twin-site studies in recent years (Yan et al., 2009; Bukowiecki et al., 2010; Gietl et al., 2010; Oliveira et al., 2010; Pey et al., 2010). With the assumption that all sources other than traffic (including any local or regional sources) have the same impact at both roadside and background sites, the increment at the roadside site obtained using the following equation is used as a local traffic increment estimate (Harrison, 2009; Yan et al., 2009; Wang et al., 2010).

$$\text{Concentration of } X_{traffic} = \text{Concentration of } X_{roadside}$$
$$- \text{Concentration of } X_{background} \tag{5}$$

Typically, the difference between rural and urban concentration observation provides an estimate for the urban increment while the difference between roadside and urban background concentration observations provides an estimate for the traffic increment (Harrison, 2009). However, results from such studies can be influenced by street geometry. Harrison et al. (2004) observed different behaviour for particle mass based on the site geometry, where the traffic contribution increases for larger daily background mass concentrations at open sites, whereas the magnitude of the traffic mass increment is independent of daily background concentration for enclosed sites, thus making selection of a suitable sampling location critical for coherent results. They also found the pattern of air circulation to be an important determinant of ambient PM concentration at an enclosed street site. Roadway studies are useful in providing a reasonable estimate of pollutant concentrations that commuters are exposed to (Phuleria et al., 2007) but if the sampling is carried out only at roadside sites, the data generated is representative only of that part of the total vehicle fleet that was sampled (Liacos et al., 2012).

Harrison et al. (2003) concluded that the trace element enrichment observed at roadside sites is contributed largely by wear and tear of tyres and brakes. Significant roadside enrichment has been reported for elements, ions and organic compounds in various cities including London (UK) (Gietl et al., 2010) Los Angeles (USA) (Ntziachristos et al., 2007), Sicily, Rome and Milan (Italy) (Manno et al., 2006; Canepari et al., 2008; Perrone et al., 2012), Barcelona (Spain) (Amato et al., 2011a,b), Portugal (Oliveira et al., 2011) and Queensland (Australia) (Gunawardana et al., 2011).. In New York City, Fe, Al and Si were found to contribute between 0.5 and 1.0 μg m^{-3} to the ambient $PM_{2.5}$ loading (Peltier et al., 2011) whereas Lin et al., 2008 reported Ca, Al and Fe to constitute between 65 and 77% of suspended particles in Taiwan. Nearly 90% of the road traffic emissions of Cu were reported to be due to brake wear in Stockholm (Sweden) (Johansson et al., 2009). Amato et al. (2011b) analyzed roadside PM_{10} enrichment (RE) of trace elements in Barcelona (Spain) reported more than 70% roadside enrichment ratio for key traffic tracers including EC, Fe, Ba, Cu, Sb,

Cr and Sn. Gietl et al. (2010) estimated that based upon published emission factors brake dust comprises nearly 13% of the total PM at a roadside site in London (UK) and Amato et al. (2011b) reported approximately 60% enrichment of brake wear particles in a road-side location compared to an urban background location in Barcelona (Spain). Pey et al. (2010) reported that the concentration of EC was found to be 3–8 times higher for the urban site in comparison with the background site in Barcelona (Spain) and Perrone et al. (2012) found EC to be contributing 11–15% of $PM_{2.5}$ in urban areas compared to 3–5% at a rural site and 1–2% at the background site in Milan (Italy). Vertical concentration gradients have also been reported for trace elements with highest concentrations at kerbside as compared with rooftop (Johansson et al., 2009). Analysis of size-resolved $PM_{2.5}$ at a roadside site in Beijing revealed that re-suspended dust and construction dust contribute 32.7–50.4% of total $PM_{2.5}$ (Song et al., 2012) and a reduction was observed in concentrations of species associated with traffic during the 2008 Olympic Games).

Comparatively higher concentrations of hopanes, steranes, *n*-alkanes and PAHs have been reported for highway/near-roadway sites compared to other sites (Yan et al., 2009; Aldabe et al., 2011; Peltier et al., 2011; Perrone et al., 2012). Yan et al. (2009) analysed $PM_{2.5}$ concentrations at urban and rural sites and observed a higher concentration for EC and primary organic compounds released by motor vehicle exhaust such as hopanes and steranes. They found organic matter to be contributing between 51 and 72% to PM2.5 mass at the urban site compared with 40–54% at the rural site. Aldabe et al. (2011) reported a concentration gradient for total carbon with the highest concentration at the roadside site and lowest concentration at a rural site, and a decreasing trend of OC concentration was observed in Prague with a decrease in traffic intensity (Ondráček et al., 2011). Gao et al. (2011) also reported higher concentrations for EC and hopanes at the roadside site compared to a rooftop site, highlighting the contribution of traffic emissions. Jones and Harrison (2006) used ratios of concentration to NO_x for determination of EFs for road traffic or particle number and mass metrics with the assumption that particle dispersion is similar to dispersion of NO_x in the atmosphere. The EFs that they obtained from this study for particle mass and number (Table 6) were comparable with emission factors calculated using other methods such as dynamometer and tunnel/roadway studies. Liacos et al. (2012) observed higher concentrations of trace metals using on-road measurements and postulated that studies using sampling from near roadside conditions might not always capture the pollutant concentrations that drivers are exposed to (Liacos et al., 2012).

5.3. NO_x as tracer

With the assumption that road traffic is the primary urban source for NO_x in the urban atmosphere, Harrison et al. (1997) used regression analysis between particle mass fractions and NO_x to obtain contributions from the traffic source for Birmingham (UK) with a total of 32% of the fine PM concentration attributed to vehicular exhaust emissions for the winter season. The method is very simple, using the assumption that the regression intercept in PM at zero NO_x is due to non-traffic sources, with the traffic contribution estimated by difference from the measured mean concentration. The same approach was also used by Fuller et al. (2002) who estimated PM_{10} and $PM_{2.5}$ concentrations using NO_x and PM emissions for London (UK) and derived a typical PM_{10}: NO_x ratio of 0.045 ± 0.003. Lim et al. (1999) also used this approach to determine the contribution of PAHs to traffic in Birmingham and concluded that traffic contributed 80–82% and 61–67% of PAH concentrations in the city centre and an urban background site (university campus) respectively. Thorpe et al. (2007) used this

method to calculate re-suspension EFs. The PM_{10} emission factor was calculated using NO_x as a tracer and coarse particle source strength and abrasion emissions were subtracted from total PM_{10} emission to a obtain a re-suspension emission factor. However, it is important to note that this method is only useful if the predominant source for local ground-level NO_x is traffic, unless used in a twin-site approach.

5.4. Other methods

Several studies have reported lower levels of vehicular emissions on weekends (Rose et al., 2006; Barmpadimos et al., 2011). Analysis of the weekly cycle of coarse PM concentrations (2.5–10 μm) with the weekly PM coarse traffic emission was carried out in Zurich (Switzerland) and it was estimated that nearly 70% and 53% of the ambient coarse PM mass is contributed by traffic on weekdays and weekends respectively (Barmpadimos et al., 2011). With the assumption that the difference in PM concentrations between weekdays and weekends at an urban background site is due to traffic movement, it is possible to calculate the traffic increment to urban PM concentrations, although the results show large uncertainties (Jones et al., 2008).

Vehicle chase experiments have also been used where vehicles are chased for a specific period of time to collect pollutant concentrations allowing data to be collected for real-world conditions of dilution (Canagaratna et al., 2010; Wang et al., 2011). However, the EFs are only valid for the specific sampling conditions and there is no control over driving conditions (Wang et al., 2011).

6. Receptor modelling of traffic PM

Watson and Chow (2007) describe receptor models as models that "interpret measurements of physical and chemical properties taken at different times and places to infer the possible sources of excessive concentrations and to quantify the contributions from those sources". A number of receptor models are used for source apportionment including the Chemical Mass Balance (CMB) model, statistical models such as PCA and PMF, Multilinear Engine (ME), Constrained Physical Receptor Model (COPREM) and UNMIX. With the assumption that the relative concentrations of chemical species are preserved between sources and receptors, receptor models use the principle of mass conservation for apportionment of PM mass to different air pollution sources. Thus, the concentration of a species measured in a particular sample can be described as (Hopke, 2003):

$$X_{ij} = \sum_{p=1}^{p} g_{ip}f_{pj} + e_{ij} \tag{6}$$

where X_{ij} is the measured concentration of the *j*th species in the *i*th sample, f_{pj} is the concentration of the *j*th species in material emitted by the source *p*, g_{ip} is the contribution of the *p*th source to the *i*th sample and e_{ij} is the portion of the measurement that cannot be fitted by the model. Different receptor models use different approaches to solve this equation, for e.g., the CMB model uses the effective-variance least squares method whereas UNMIX uses eigenvector analysis. Concentrations of trace elements (e.g. Si, Fe, Cu), ions (e.g. SO_4^{2-}), EC/OC and organic compounds (e.g. PAHs, hopanes, alkanoic acids) are used as inputs for the models.

In recent years, there have been many studies for source apportionment of road traffic-generated PM, some of which appear in Table 7. There have also been studies that have used both CMB and factor analysis models (Larsen and Baker, 2003; Bullock et al., 2008).

Table 7
Source apportionment analyses for road traffic-generated PM (selected studies).

Reference	Country	Method	Particle type	Sources
Lin et al. (2005)	Taiwan	PCA	Coarse particles Fine particles Ultrafine and nano particles	Fugitive dust and brake lining, diesel, fuel oil, gasoline Gasoline, diesel, brake lining/tyre wear, fuel oil Gasoline, diesel, industry, fuel oil
Furusjo et al. (2007)	Stockholm, Sweden	PMF	PM_{10}	Re-suspension Vehicle derived (brake wear) Road salt Regional combustion Long-range transport
Chan and Mozurkewich (2007)	Ontario, Canada	Absolute PCA	Particle number distribution with trace gas and meteorological data	Common sources across 3 sites: Nucleation particles (photochemically produced), regional pollution factor, boundary layer dynamics factor Specific sources at the three sites: Local industrial emissions (urban) Processed nucleation mode particles (polluted rural) Transported fine particles (downwind Toronto)
Amato et al. (2009)	Barcelona, Spain	PMF2 with ME	PM_{10} and $PM_{2.5}$	Road dust factor accounted for 17% PM10, 8% $PM_{2.5}$ and less than 2% of PM_1. All traffic sources contributed to 46% PM10, 51% $PM_{2.5}$ and 48% of PM_1.
Fabretti et al. (2009)	Nice, France	PMF	$PM_{2.5}$	Vehicle abrasion: Cu, Zn, Sb (36%) Re-suspension: Mn, Fe, As, Rb, Sr (43%) Fuel combustion: V, Ni, Co (21%)
Dreyfus et al. (2009)	Wilmington, USA	PMF	Organic carbon	Diesel factor: Alkyl fragment, benzothiazole, benzenedicarboxylic acids/ 3,4-dimethoxybenzaldehyde, 3,4-dimethoxybenzoic acid, hopanes Road dust/car emissions: hopanes, benzothiazole and triacontanoic acid
Mancilla and Mendoza (2012)	Monterrey, Mexico	FA	$PM_{2.5}$	Brake emissions Exhaust emissions
Ondráček et al. (2011)	Prague, Czech Republic			Abrasion of vehicle parts: Fe, Cu, Mn, Zn Re-suspension of road dust: Si, Al, Ca Long-range transport/regional background: Ca, K
Song and Gao (2011)	New Jersey, USA	FA	PM	Brake wear and fuel combustion: Fe, Sb, Pb, Cd ($\sim$35%) Primary fuel combustion: Cr, Ni, V, Cu ($\sim$28.3%) Tyre wear and fuel combustion: Zn, Co ($\sim$23.7%)
Yin et al. (2010)	United Kingdom	CMB	$PM_{2.5}$	Vegetative detritus; woodsmoke; natural gas; diesel engines (11.6%); gasoline engines (1.4%); smoking engines (9.7%); coal; dust/soil; secondary organic matter; sea salt; ammonium sulphate; ammonium nitrate

6.1. Multivariate statistical methods

There are a number of different methods based on factor analysis including PCA, PMF, UNMIX and ME. Such methods do not require a priori information about source emission characteristics and are useful in cases where relevant source profiles are not available (Hopke, 2003; Viana et al., 2008). In the simplest matrix form, the equation (6) can be represented as (Hopke, 2003)

$$X = GF'$$

A number of studies have been conducted using the different factor analysis methods and studies in different locations have often reported different magnitudes of exhaust and non-exhaust emissions. In Taiwan, gasoline, diesel and fuel oil emissions together with dust and brake wear were found to be key contributors to traffic-generated PM using PCA (Lin et al., 2005). Using COPREM, Wahlin et al. (2006) apportioned the local traffic emissions to exhaust, road dust, brake wear and winter salting in Copenhagen (Denmark). Amato et al. (2009) analysed road dust in Barcelona (Spain) using ME-2, and re-suspension was found to contribute 37%, 15% and 3% of total traffic emissions respectively in PM_{10}, $PM_{2.5}$ and PM_1 respectively. Other identified sources included secondary aerosols and a soil source. Sjodin et al. (2010) found road wear to be the most important source of PM_{10} emissions in streets as well as background locations in Sweden, which is probably a reflection of road sanding and use of studded tyres. Using PMF, three distinct traffic sources including vehicle abrasion (36%), re-suspension (43%) and fuel combustion (21%) were resolved for traffic sources using trace metals in Nice (France) from data

collected in a road tunnel (Fabretti et al., 2009). Based on PMF analysis in Switzerland, Bukowiecki et al. (2010) reported 21%, 38% and 41% of the traffic emissions to be contributed by brake wear, re-suspended road dust and exhaust emissions respectively for street canyons. For freeways, while the contribution of exhaust emissions remained the same, the contribution of brake wear was much less at 3% whereas re-suspended road dust contributed as much as 56%. Movement of traffic also contributes to re-suspension of road dust and re-suspended road dust has been estimated to contribute as much as 38% of the total PM in a street canyon and 56% on a highway (Bukowiecki et al., 2010; Laidlaw et al., 2012). Analysis of PM in Madrid (Spain) using PMF estimated that nearly 29% of the PM_{10} in the city is contributed by road dust while another 31% is contributed by direct vehicular emission (Karanasiou et al., 2011). In Oporto (Portugal), exhaust emissions and re-suspension due to traffic were found to contribute 12–55% of the coarse PM and 14–66% of the fine PM based on PCA- multiple regression analysis and inter-site mass balance analysis (Oliveira et al., 2010). They used inter-site mass balance for separating the traffic increment in the road dust and concluded that nearly 30% of the fine fraction and 50–70% of the coarse fraction PM is contributed by road dust. They also used a roadside impact factor (defined as the fractional difference between the concentrations at the roadside and urban background) and the highest factors were observed for Cu, Zr, Fe, Cr and Sn indicating a traffic source for these metals. Thorpe et al. (2007) reported re-suspended road dust, abrasion source emissions and exhaust and fine range abrasion emissions to contribute 20–22%, 19% and 60% to roadside incremental PM_{10} concentration respectively in a major highway in London. Four major sources for road surface dust particles of diameter less than 10 μm including

motor exhaust, brake and tyre wear and road surface wear together with background urban dust were reported for European cities with an increase in the concentrations of trace metals and OC/EC with an increase in traffic volume (Amato et al., 2011a). Song and Gao (2011) identified three distinct sources of trace metals at a highway site in New Jersey (USA) using factor analysis: brake wear and fuel combustion, primary fuel combustion and tyre wear and fuel combustion. PCA analysis was used for determining PM$_{2.5}$ sources in Incheon (Korea) and motor vehicles/sea salt and soil emissions were found to contribute 39% and 7% respectively (Choi et al., 2012). Similar results were obtained by Cheng et al. (2011) in Hong Kong where traffic emissions and re-suspended dust emissions were found to contribute 38% and 5% using PMF.

Dreyfus et al. (2009) conducted PMF analysis with time-resolved organic molecular marker data and identified diesel and road dust/car emissions factors which accounted for 2/3rd of the total OC concentration. Receptor modelling analysis for PM$_{2.5}$ was conducted using PMF and UNMIX, and PMF distinguished between diesel and gasoline engine emissions using temperature resolved carbon fractions (Sahu et al., 2011). Pey et al. (2009) performed source apportionment using PCA and regression with a large dataset including information on particle number concentration; size distribution and chemical composition for an urban background site in Barcelona, and road traffic was identified as the most significant source (52–86%) for aerosols in the sub-micrometer range.

Liu et al. (2007) attributed the PAHs in road dust in Shanghai (China) to traffic emissions and coal combustion using principal component analysis (PCA) and compositional analysis. Using the same technique, Han et al. (2009) attributed PAHs in road dust in Anshan (China) to emissions from traffic, coal combustion, steel industry and cooking. Using MLR, Boonyatumanond et al. (2007) reported tyres, diesel vehicle soot and used engine oil to be the key sources for PAHs found in street dust in Bangkok (Thailand). Hassanien and Abdel-Latif (2008) attributed PAHs in road dust in Cairo (Egypt) to emissions from traffic, industry and incomplete combustion of waste based on correlation analysis. Teixiera et al. (2012) reported vehicular emissions and coal and wood combustion to be the key sources for PAHs in Porte Alegre (Brazil) using PCA.

Other approaches have also been used for estimation of source contributions. Recently, Harrison et al. (2012b) used field data for estimating the contribution of brake wear, tyre wear and road dust using size distributions of Ba, Zn and Si as the tracers for the sources respectively. By merging size distributions measured with a Scanning Mobility Particle Sizer (SMPS) and an Aerodynamic Particle Sizer (APS), Harrison et al. (2011) were able to generate continuous particle size distributions for a roadside site from 15 nm–10 μm diameter. By applying PMF, they were able to separately identify nucleation mode and accumulation mode exhaust particles as well as brake dust and re-suspension particles, and several particle classes arising from the urban background.

6.2. Chemical mass balance (CMB) model

The CMB model uses the ambient measurement data for chemical species together with the associated uncertainty and source profiles for different sources as inputs and the output consists of estimates of contribution of each source to the total mass. The CMB model has several assumptions including (Watson et al., 2002):

- chemical species do not react with one another and the species composition does not change between the source(s) and the receptor(s);
- number of species are more than the number of sources;

- all of the potential sources are included the model;
- source profiles are not collinear;
- random and non-correlated uncertainties.

It is important to note that while the multivariate models do not require any information on source emission composition, the CMB model requires the species abundances (in the form of source profiles) in each source as a model input. Source profiles are "the mass abundances, i.e. fraction of total mass of chemical species in source emissions, and such profiles are generally representative of source categories rather than individual emitters" (Watson et al., 2002). There are several ways of preparing source profiles for road vehicles including tests with chassis dynamometers, measurements in tunnels/highways and near roadside measurements (Phuleria et al., 2007; Handler et al., 2008; Yan et al., 2009) as discussed in section 6. Source profiles are used for quantification of source contributions to particulate matter using the chemical mass balance (CMB) model as well as to compare and validate results obtained from factor analysis models. The CMB model relies to a large extent on the accuracy of the source profiles used as an input. The typical components of any source profile are found to be more-or-less similar although the mass abundances vary depending on location and emitter characteristics. Thus, different combinations of source profiles can provide statistically valid yet completely different solutions and selection of a source profile comparable to ambient data is one of the key prerequisites for a good CMB solution (Robinson et al., 2006). Absence of locally-representative source profiles and inter-region variability can result in errors in estimation of source contributions from the CMB model (Yan et al., 2009; Rutter et al., 2011). Significant differences have also been reported between real-world emissions and laboratory-tested emissions (Yan et al., 2009; Ancelet et al., 2011). Analysis of vehicular source profiles in Bangkok, Thailand led to the conclusion that while source profiles for different vehicle categories vary significantly, ranges of the light duty and heavy duty average source profiles overlap and can be represented by a single composite profile (Oanh et al., 2010). Fraser et al. (2003) and Fujita et al. (2007) used the CMB model to analyse the apportionment of PM mass between diesel and gasoline vehicles. While high molecular weight (4–7 ring) PAHs (coronene, benzo(ghi)perylene), hopanes and steranes were found to be influential in case of gasoline engine emissions, 2–4 ring methylated PAHs and EC were reported to be key species in the case of diesel engine emissions. Hanedar et al. (2008) used the CMB model to resolve the sources of PAHs in Istanbul (Turkey) and identified vehicle emissions (gasoline and diesel), natural gas combustion, coal and wood burning as major sources with vehicular emissions contributing more than 50% at both urban and rural location. Perrone et al. (2012) used CMB analysis to resolve PM sources in Milan (Italy) and reported traffic emissions to contribute between 17 and 24% to PM$_{2.5}$ mass in an urban location compared to 7–9% at a rural site and El Haddad et al. (2011) reported vehicular emissions to contribute 17% to the PM mass in Marseille (France). In Detroit (USA), traffic emissions and road dust emissions were reported to contribute 24–36% and 3–5% respectively of mass using CMB analysis of PM$_{2.5}$ data.

In recent years, there have been several studies focussing on characterization of trace metals and/or organic marker compounds in PM contributed by road traffic emissions and preparation of mixed traffic source profiles (El Haddad et al., 2009; Fabretti et al., 2009; Yan et al., 2009; Ancelet et al., 2011). El Haddad et al. (2009) prepared a source profile for the vehicle fleet in France using tunnel measurements and nearly 70 organic markers, whereas Fabretti et al. (2009) used trace element marker data from tunnel-based measurements to derive chemical fingerprints for different traffic-related sources including combustion, re-suspended dust

and abrasion-related particles. Tunnel studies were also used in USA for preparation of source profiles for gasoline and diesel engines using inorganic and organic markers (Phuleria et al., 2006; Landis et al., 2007). Yin et al. (2010) used organic molecular source tracers to provide separate quantification of gasoline, diesel and smoker vehicle emissions at two UK sites. Recently, an attempt has also been made to characterize the contribution of lubricating oil and fuel to the PM emissions (Sonntag et al., 2012).

7. Conclusions

Road traffic frequently makes a very significant contribution (5–80% depending on site and location) to airborne concentrations of PM, whether expressed in terms of particle mass or number concentration (Thorpe and Harrison, 2008; Belis et al., 2013). Particles arise not only from the engine exhaust but also from abrasion of tyres, road surface and brake components as well as from re-suspension from the road surface. The sizes of particles emitted from these sources differ considerably and size association may be a useful guide to particle source, as is chemical composition. Fleet mix, type of road, traffic characteristic and fuel type are the factors that impact the emission type and the relative contributions from exhaust and non-exhaust emissions. The determination of particle mass and number increments due to traffic emissions can be based on a number of methods including measurements in well controlled environments such as a road tunnel, or twin site studies such as a highway and urban background (allowing for subtraction of the background contribution), as well as using vehicle-specific tracers and the more sophisticated receptor modelling methods based on multivariate statistics or Chemical Mass Balance which have been used in many studies to estimate the contributions to particulate matter concentrations, not only from road vehicles but also from other sources. However, significant gaps still exist in our understanding of traffic emissions, particularly:

1. Characterization and quantification of different non-exhaust sources.
2. Impacts of non-exhaust emissions upon human health: While there are many studies reporting links between exposure to air pollution and adverse health impacts, detailed information on the components that contribute to toxicity is missing (Kelly and Fussell, 2012).
3. Aerosol chemistry in high traffic environments, and the evolution of particles emitted from vehicles.

Further, most of the reported analyses have been carried out in the USA or Europe, and there is a lack of reliable information on traffic emissions in areas with high population density in Africa, Asia and South America. The field offers much scope for future research, including the development of enhanced methods for quantification of non-exhaust contributions to airborne concentrations.

References

Aatmeeyata, K.D.S., Sharma, M., 2009. Traffic generated non-exhaust particulate emissions from concrete pavement: a mass and particle size study for two-wheelers and small cars. Atmospheric Environment 43 (35), 5691–5697.

Aatmeeyata, K.D.S., Sharma, M., 2010. Polycyclic aromatic hydrocarbons, elemental and organic carbon emissions from tire-wear. Science of the Total Environment 408 (20), 4563–4568.

Abu-Allaban, M., Coulomb, W., Gertler, A.W., Gillies, J., Pierson, W.R., Rogers, C.F., Sagebiel, J.C., Tarnay, L., 2002. Exhaust particle size distribution measurements at the Tuscarora Mountain Tunnel. Aerosol Science and Technology 36 (6), 771–789.

Abu-Allaban, M., Gillies, J.A., Gertler, A.W., Clayton, R., Proffitt, D., 2003. Tailpipe, resuspended road dust, and brake wear emission factors from on-road vehicles. Atmospheric Environment 37, 5283–5293.

Adachi, K., Tainosho, Y., 2004. Characterization of heavy metal particles embedded in tire dust. Environment International 30 (8), 1009–1017.

Aldabe, J., Elustondo, D., Santamaría, C., Lasheras, E., Pandolfi, M., Alastuey, A., Querol, X., 2011. Chemical characterisation and source apportionment of PM2.5 and PM10 at rural, urban and traffic sites in Navarra (North of Spain). Atmospheric Research 102 (1–2), 191–205.

Allen, A.G., Nemitz, E., Shi, J.P., Harrison, R.M., Greenwood, J.C., 2001. Size distributions of trace metals in atmospheric aerosols in the United Kingdom. Atmospheric Environment 35 (27), 4581–4591.

Allen, J.O., Alexandrova, O., Kaloush, K.E., 2006. Tire Wear Emissions for Asphalt Rubber and Portland Cement Concrete Pavement Surfaces. Arizona State University. Report submitted to Arizona Department of Transportation (Contract KR-04-0720-TRN).

Almeida, S.M., Pio, C.A., Freitas, M.C., Reis, M.A., Trancoso, M.A., 2006. Source apportionment of atmospheric urban aerosol based on weekdays/weekend variability: evaluation of road re-suspended dust contribution. Atmospheric Environment 40 (11), 2058–2067.

Amato, F., Pandolfi, M., Viana, M., Querol, X., Alastuey, A., Moreno, T., 2009. Spatial and chemical patterns of PM10 in road dust deposited in urban environment. Atmospheric Environment 43 (9), 1650–1659.

Amato, F., Pandolfi, M., Moreno, T., Furger, M., Pey, J., Alastuey, A., Bukowiecki, N., Prevot, A.S.H., Baltensperger, U., Querol, X., 2011a. Sources and variability of inhalable road dust particles in three European cities. Atmospheric Environment 45 (37), 6777–6787.

Amato, F., Viana, M., Richard, A., Furger, M., Prévôt, A.S.H., Nava, S., Lucarelli, F., Bukowiecki, N., Alastuey, A., Reche, C., Moreno, T., Pandolfi, M., Pey, J., Querol, X., 2011b. Size and time-resolved roadside enrichment of atmospheric particulate pollutants. Atmospheric Chemistry and Physics 11 (6), 2917–2931.

Amato, F., Karanasiou, A., Moreno, T., Alastuey, A., Orza, J.A.G., Lumbreras, J., Borge, R., Boldo, E., Linares, C., Querol, X., 2012. Emission factors from road dust resuspension in a Mediterranean freeway. Atmospheric Environment 61, 580–587.

Ancelet, T., Davy, P.K., Trompetter, W.J., Markwitz, A., Weatherburn, D.C., 2011. Carbonaceous aerosols in an urban tunnel. Atmospheric Environment 45 (26), 4463–4469.

Apeagyei, E., Bank, M.S., Spengler, J.D., 2011. Distribution of heavy metals in road dust along an urban–rural gradient in Massachusetts. Atmospheric Environment 45 (13), 2310–2323.

Ban-Weiss, G.A., Lunden, M.M., Kirchstetter, T.W., Harley, R.A., 2010. Size-resolved particle number and volume emission factors for gasoline and diesel motor vehicles. Aerosol Science 41, 5–12.

Barmpadimos, I., Nufer, M., Oderbolz, D.C., Keller, J., Aksoyoglu, S., Hueglin, C., Baltensperger, U., Prevot, A.S.H., 2011. The weekly cycle of ambient concentrations and traffic emissions of coarse (PM10–PM2.5) atmospheric particles. Atmospheric Environment 45 (27), 4580–4590.

Beddows, D.C.S., Harrison, R.M., 2008. Comparison of average particle number emission factors for heavy and light duty vehicles derived from rolling chassis dynamometer and field studies. Atmospheric Environment 42, 7954–7966.

Beddows, D.C.S., Dall'Osto, M., Harrison, R.M., 2009. Cluster analysis of rural, urban and curbside atmospheric particle size data. Environmental Science and Technology 43, 4694–4700.

Belis, C.A., Karagulian, F., Larsen, B.R., Hopke, P.K., 2013. Critical review and meta-analysis of ambient particulate matter source apportionment using receptor models in Europe. Atmospheric Environment 69, 94–108.

Berger, J., Denby, B., 2011. A generalized model for traffic induced road dust emissions. Model description and evaluation. Atmospheric Environment 45, 3692–3703.

Bhaskar, V.S., Sharma, M., 2008. Assessment of fugitive road dust emissions in Kanpur, India. Transportation Research Part D 13, 400–403.

Birmili, W., Allen, A.G., Bary, F., Harrison, R.M., 2006. Trace metal concentrations and water solubility in size-fractionated atmospheric particles and influence of road traffic. Environmental Science and Technology 40 (4), 1144–1153.

Blau, P.J., Meyer III, H.M., 2003. Characteristics of wear particles produced during friction tests of conventional and unconventional disc brake materials. Wear 255, 1261–1269.

Boonyatumanond, R., Murakami, M., Wattayakorn, G., Togo, A., Takeda, H., 2007. Sources of polycyclic aromatic hydrocarbons (PAHs) in street dust in a tropical Asian mega-city, Bangkok, Thailand. Science of the Total Environment 384, 420–432.

Boulter, P.G., Thorpe, A.J., Harrison, R.M., Allen, A.G., 2005. Road Vehicle Non-exhaust Particulate Matter: Final Report on Emission Modelling. Published Project Report PPR110. Available at: http://uk-air.defra.gov.uk/reports/cat15/0706061624_Report2__Emission_modelling.PDF.

Brook, J.R., Graham, L., Charland, J.P., Cheng, Y., Fan, X., Lu, G., Li, S.M., Lillyman, C., MacDonald, P., Caravaggio, G., MacPhee, J.A., 2007. Investigation of the motor vehicle exhaust contribution to primary fine particle organic carbon in urban air. Atmospheric Environment 41 (1), 119–135.

Buckeridge, D.L., Glazier, R., Harvey, B.J., Escobar, M., Amrhein, C., Frank, J., 2002. Effect of motor vehicle emissions on respiratory health in an urban area. Environmental Health Perspectives 110 (3), 293–300.

Bukowiecki, N., Lienemann, P., Hill, M., Furger, M., Richard, A., Amato, F., Prevot, A.S.H., Baltensperger, U., Buchmann, B., Gehrig, R., 2010. PM10 emission factors for non-exhaust particles generated by road traffic in an urban street canyon and along a freeway in Switzerland. Atmospheric Environment 44 (19), 2330–2340.

Bullock, K.R., Duvall, R.M., Norris, G.A., McDow, S.R., Hays, M.D., 2008. Evaluation of the CMB and PMF models using organic molecular markers in fine particulate matter collected during the Pittsburgh Air Quality Study. Atmospheric Environment 42, 6897—6904.

Caceres, D., Adonis, M., Gil, L., Miguel, A.H., 1998. Exhaust PAH in Chilean vehicles: levels, source signatures and receptor modeling. Journal of Aerosol Science 29 (Suppl. 2(0)), S741—S742.

Camatani, M., Crosta, G.F., Dolukhanyan, T., Sung, C., Giuliani, G., Corbetta, G.M., Cencetti, S., Regazzoni, C., 2001. Microcharacterization and identification of tire debris in heterogenous laboratory and environmental specimens. Materials Characterization 46, 271—283.

Canagaratna, M.R., Onasch, T.B., Wood, E.C., Herndon, S.C., Jayne, J.T., Cross, E.S., Miake-Lye, R.C., Kolb, C.E., Worsnop, D.R., 2010. Evolution of vehicle exhaust particles in the Atmosphere. Journal of Air and Waste Management Association 60, 1192—1203.

Canepari, S., Perrino, C., Olivieri, F., Astolfi, M.L., 2008. Characterisation of the traffic sources of PM through size-segregated sampling, sequential leaching and ICP analysis. Atmospheric Environment 42 (35), 8161—8175.

Cass, G.R., 1998. Organic molecular tracers for particulate air pollution sources. Trends in Analytical Chemistry 17 (6), 356—366.

Chan, D., Stachowiak, G.W., 2004. Review of automotive brake friction materials. Proceedings of the Institution of Mechanical Engineers, Part D: Journal of Automobile Engineering 218 (9), 953—966.

Chan, T.W., Mozurkewich, M., 2007. Application of absolute principal component analysis to size distribution data: identification of particle origins. Atmospheric Chemistry and Physics 7, 887—897.

Chang, T.Y., Modzelewski, S.W., Norbeck, J.M., Pierson, W.R., 1981. Tunnel air quality and vehicle emissions. Atmospheric Environment 15 (6), 1011—1016.

Charron, A., Harrison, R.M., 2003. Primary particle formation from vehicle emissions during exhaust dilution in the roadside atmosphere. Atmospheric Environment 37 (29), 4109—4119.

Charron, A., Harrison, R.M., 2009. Atmospheric Nanoparticles. Environmental and Human Health Impacts of Nanotechnology. John Wiley and Sons, Ltd., pp. 63—209.

Chellam, S., Kulkarni, P., Fraser, M.P., 2005. Emissions of organic compounds and trace metals in fine particulate matter from motor vehicles: a tunnel study in Houston, Texas. Journal of Air and Waste Management Association 55, 60—72.

Chen, J., Zheng, H., Wang, W., Liu, H., Lu, L., Bao, L., Ren, L., 2006. Resuspension method for road surface dust collection and aerodynamic size distribution characterization. China Particuology 4 (6), 300—303.

Chen, F., Hu, W., Zhong, Q., 2013. Emissions of particle-phase polycyclic aromatic hydrocarbons (PAHs) in the Fu Gui-Shan Tunnel of Nanjing, China. Atmospheric Research 124, 53—60.

Cheng, Y., Lee, S.C., Ho, K.F., Chow, J.C., Watson, J.G., Louie, P.K.K., Cao, J.J., Hai, X., 2010. Chemically-speciated on-road PM2.5 motor vehicle emission factors in Hong Kong. Science of the Total Environment 408 (7), 1621—1627.

Cheng, Y., Zou, S.C., Lee, S.C., Chow, J.C., Ho, K.F., Watson, J.G., Han, Y.M., Zhang, R.J., Zhang, F., Yau, P.S., Huang, Y., Bai, Y., Wu, W.J., 2011. Characteristics and source apportionment of PM1 emissions at a roadside station. Journal of Hazardous Materials 195, 82—91.

Chiang, H.-L., Huang, Y.-S., 2009. Particulate matter emissions from on-road vehicles in a freeway tunnel study. Atmospheric Environment 43, 4014—4022.

Chiang, H.-L., Lai, Y.-M., Chang, S.-Y., 2012. Pollutant constituents of exhaust emitted from light-duty diesel vehicles. Atmospheric Environment 47, 399—406.

Chirico, R., Prevot, A.S.H., DeCarlo, P.F., Heringa, M.F., Richter, R., Weingartner, E., Baltensperger, U., 2011. Aerosol and trace vehicle emission factors measured in a tunnel using an aerosol mass spectrometer and other on-line instrumentation. Atmospheric Environment 45, 2182—2192.

Choi, J.-K., Heo, J.-B., Ban, S.-J., Yi, S.-M., Zoh, K.-D., 2012. Chemical characteristics of PM2.5 aerosol in Incheon, Korea. Atmospheric Environment 60, 583—592.

Councell, T.B., Duckenfield, K.U., Landa, E.R., Callender, E., 2004. Tire wear particles as a source of Zn to the environment. Environmental Science and Technology 38 (15), 4206—4214.

Crawford, J., Chambers, S., Cohen, D.D., Dyer, L., Wang, T., Zahorowski, W., 2007. Receptor modelling using positive matrix factorisation, back trajectories and Radon-222. Atmospheric Environment 41 (32), 6823—6837.

Cyrys, J., Pitz, M., Heinrich, J., Wichmann, H.E., Peters, A., 2008. Spatial and temporal variation of particle number concentration in Augsburg, Germany. Science of the Total Environment 401 (1—3), 168—175.

Dahl, A., Gharibi, A., Swietlicki, E., Gudmundsson, A., Bohgard, M., Ljungman, A., Blomqvist, G., Gustafsson, M., 2006. Traffic-generated emissions of ultrafine particles from pavement—tire interface. Atmospheric Environment 40 (7), 1314—1323.

Dall'Osto, M., Thorpe, A., Beddows, D.C.S., Harrison, R.M., Barlow, J.F., Dunbar, T., Williams, P.I., Coe, H., 2011. Remarkable dynamics of nanoparticles in the urban atmosphere. Atmospheric Physics and Chemistry 11, 6623—6637.

Denier van der Gon, H., Gerlofs-Nijland, M.E., Gehrig, R., Gustafsson, M., Janssen, N., Harrison, R.M., Hulskotte, J., Johansson, C., Jozwicka, M., Keuken, M., Krijgsheld, K., Ntziachristos, L., Riediker, M., Cassee, F.R., 2013. The policy relevance of wear emissions from road transport, now and in the future — an international workshop report and consensus statement. Journal of the Air and Waste Management Association 63 (2), 136—149.

Dongarra, G., Manno, E., Varrica, D., 2009. Possible markers of traffic-related emissions. Environmental Monitoring and Assessment 154, 117—125.

Dreyfus, M.A., Adou, K., Zucker, S.M., Johnston, M.V., 2009. Organic aerosol source apportionment from highly time-resolved molecular composition measurements. Atmospheric Environment 43 (18), 2901—2910.

Duong, T.T., Lee, B.K., 2011. Determining contamination level of heavy metals in road dust from busy traffic areas with different characteristics. Journal of Environmental Management 92 (3), 554—562.

El Haddad, I., Marchand, N., Dron, J., Temime-Roussel, B., Quivet, E., Wortham, H., Jaffrezo, J.L., Baduel, C., Voisin, D., Besombes, J.L., Gille, G., 2009. Comprehensive primary particulate organic characterization of vehicular exhaust emissions in France. Atmospheric Environment 43 (39), 6190—6198.

El Haddad, I., Marchand, N., Wortham, H., Piot, C., Besombes, J.-L., Cozic, J., Chauvel, C., Armengaud, A., Robin, D., Jaffrezo, J.-L., 2011. Primary sources of PM2.5 organic aerosol in an industrial Mediterranean city, Marseille. Atmospheric Chemistry and Physics 11, 2039—2058.

Etyemezian, V., Kuhns, H., Gillies, J., Chow, J., Hendrickson, K., McGown, M., Pitchford, M., 2003. Vehicle-based road dust emissions measurement (III): effect of speed, traffic volume, location, and season on PM10 road dust emissions in the Treasure Valley, ID. Atmospheric Environment 37, 4583—4593.

Fabretti, J.-F., Sauret, N., Gal, J.-F., Maria, P.-C., Schärer, U., 2009. Elemental characterization and source identification of PM2.5 using positive matrix factorization: the Malraux road tunnel, Nice, France. Atmospheric Research 94 (2), 320—329.

Fan, Y., Meng, Q., Weisel, C., Shalat, S., Laumbach, R., Ohman-Strickland, P., Black, K., Rodriguez, M., Bonanno, L., 2006. Acute short-term exposures to PM2.5 generated by vehicular emissions and cardiopulmonary effects in older adults. Epidemiology 17 (6), S213—S214.

Fang, G.-C., Chang, C.-N., Wu, Y.-S., Fu, P.P.-C., Yan, I.-L., Chen, M.-H., 2004. Characterization, identification of ambient air and road dust polycyclic aromatic hydrocarbons in central Taiwan, Taichung. Science of the Total Environment 327, 135—146.

Franco, V., Kousoulidou, M., Muntean, M., Ntziachristos, L., Hausberger, S., Dilara, P., 2013. Road vehicle emission factors development: a review. Atmospheric Environment 70, 84—97.

Fraser, M.P., Buzcu, B., Yue, Z.W., Mcgaughey, G.R., Desai, N.R., Allen, D.T., Seila, R.L., Lonneman, W.A., Harley, R.A., 2003. Separation of fine particulate matter emitted from gasoline and diesel vehicles using chemical mass balancing techniques. Environmental Science and Technology 37, 3904—3909.

Fraser, M.P., Cass, G.R., Simoneit, B.R.T., 1998. Gas-phase and particle-phase organic compounds emitted from motor vehicle traffic in a Los Angeles roadway tunnel. Environmental Science and Technology 32, 2051—2060.

Fujita, E.M., Campbell, D.E., Zielinska, B., 2006. Chemical Analysis of Lubrication Oil Samples from a Study to Characterize Exhaust Emissions from Light-duty Gasoline Vehicles in the Kansas City Metropolitan Area. (Final Report).

Fujita, E.M., Campbell, D.E., Arnott, W.P., Chow, J.C., Zielinska, B., 2007. Evaluations of the chemical mass balance method for determining contributions of gasoline and diesel exhaust to ambient carbonaceous aerosols. Journal of Air and Waste Management Association 57, 721—740.

Fuller, G.W., Carslaw, D.C., Lodge, H.W., 2002. An empirical approach for the prediction of daily mean PM10 concentrations. Atmospheric Environment 36, 1431—1441.

Fulper, C.R., Kishan, S., Baldauf, R.W., Sabisch, R., Warila, J., Fujita, E.M., Scarbro, C., Crews, W.S., Snow, R., Gabele, P., Sanots, R., Tierney, E., Cantrell, B., 2010. Methods of characterizing the distribution of exhaust emissions from light-duty, gasoline-powered motor vehicles in the U.S. Fleet. Journal of Air and Waste Management Association 60, 1376—1387.

Funasaka, K., Miyazaki, T., Kawaraya, T., Tsuruho, K., Mizuno, T., 1998. Characteristics of particulates and gaseous pollutants in a highway tunnel. Environmental Pollution 102 (2—3), 171—176.

Furusjo, E., Sternbeck, J., Cousins, A.P., 2007. PM10 source characterization at urban and highway roadside locations. Science of the Total Environment 387, 206—219.

Fushimi, A., Hasegawa, S., Takahashi, K., Fujitani, Y., Tanabe, K., Kobayashi, S., 2008. Atmospheric fate of nuclei-mode particles estimated from the number concentrations and chemical composition of particles measured at roadside and background sites. Atmospheric Environment 42 (5), 949—959.

Gao, B., Yu, J.-Z., Li, S.-X., Ding, X., He, Q.-F., Wang, X.-M., 2011. Roadside and rooftop measurements of polycyclic aromatic hydrocarbons in PM2.5 in urban Guangzhou: evaluation of vehicular and regional combustion source contributions. Atmospheric Environment 45 (39), 7184—7191.

Garg, B.D., Cadle, S.H., Mulawa, P.A., Groblicki, P.J., Laroo, C., Parr, G.A., 2000. Brake wear particulate matter emissions. Environmental Science and Technology 34 (21), 4463—4469.

Gehrig, R., Hill, M., Buchmann, B., Imhof, D., Weingartner, E., Baltensperger, U., 2004. Separate determination of PM10 emission factors of road traffic for tailpipe emissions and emissions from abrasion and resuspension processes. International Journal of Environment and Pollution 22 (3), 312—325.

Gehrig, M., Zeyer, K., Bukowiecki, N., Lienemann, P., Poulikakos, L.D., Furger, M., Buchmann, B., 2010. Mobile load simulators — a tool to distinguish between the emissions due to abrasion and resuspension of PM10 from road surfaces. Atmospheric Environment 44, 4937—4943.

Gertler, A.W., Pierson, W.R., 1996. Recent measurements of mobile source emission factors in North American tunnels. Science of the Total Environment 189/190, 107—113.

Gertler, A.W., Gillies, J.A., Pierson, W.R., Rogers, C.F., Sagebiel, J.C., Abu-Allaban, M., Coulombe, W., Tarnay, L., Cahill, T.A., 2002. Real-world Particulate Matter and

Gaseous Emissions from Motor Vehicles in a Highway Tunnel. Research Report 107. Health Effects Institute.

Gertler, A., Kuhns, H., Abu-Allaban, M., Damm, C., Gilies, J., Etyemezian, V., Clayton, R., Proffitt, D., 2006. A case study of the impact of winter road sand/salt and street sweeping on road dust re-entrainment. Atmospheric Environment 40, 5976–5985.

Gietl, J.K., Lawrence, R., Thorpe, A.J., Harrison, R.M., 2010. Identification of brake wear particles and derivation of a quantitative tracer for brake dust at a major road. Atmospheric Environment 44 (2), 141–146.

Gillies, J.A., Gertler, A.W., Sagebiel, J.C., Dippel, W.A., 2001. On-road particulate matter (PM2.5 and PM10) emissions in the Sepulveda tunnel, Los Angeles, California. Environmental Science and Technology 35, 1054–1063.

Grieshop, A.P., Lipsky, E.M., Pekney, N.J., Takahama, S., Robinson, A.L., 2006. Fine particle emission factors from vehicles in a highway tunnel: effects of fleet composition and season. Atmospheric Environment 40, 287–298.

Gunawardana, C., Goonetilleke, A., Egodawatta, P., Dawes, L., Kokot, S., 2011. Source characterisation of road dust based on chemical and mineralogical composition. Chemosphere 87 (2), 163–170.

Gustafsson, M., Blomqvist, G., Gudmundsson, A., Dahl, A., Swietlicki, E., Bohgard, M., Lindbom, J., Ljungman, A., 2008. Properties and toxicological effects of particles from the interaction between tyres, road pavement and winter traction material. Science of the Total Environment 393 (2–3), 226–240.

Gustafsson, M., Blomqvist, A., Gudmundsson, A., Dahl, A., Jonsson, P., Swietlicki, E., 2009. Factors influencing PM10 emissions from road pavement wear. Atmospheric Environment 49, 226–240.

Han, L., Zhuang, G., Cheng, S., Wang, Y., Li, J., 2007. Characteristics of re-suspended road dust and its impact on the atmospheric environment in Beijing. Atmospheric Environment 41 (35), 7485–7499.

Han, B., Bai, Z., Guo, G., Wang, F., Li, F., Liu, Q., Ji, Y., Li, X., Hu, Y., 2009. Characterization of PM(10) fraction of road dust for polycyclic aromatic hydrocarbons (PAHs) from Anshan, China. Journal of Hazardous Materials 170 (2–3), 934–940.

Handler, M., Puls, C., Zbiral, J., Marr, I., Puxbaum, H., Limbeck, A., 2008. Size and composition of particulate emissions from motor vehicles in the Kaisermühlen-Tunnel, Vienna. Atmospheric Environment 42 (9), 2173–2186.

Hanedar, A., Alp, K., Kaynak, B., Baek, J., Avsar, E., Odman, M.T., 2008. Concentrations and sources of PAHs at three stations in Istanbul, Turkey. Atmospheric Research 99, 391–399.

Harris, S.J., Maricq, M.M., 2001. Signature size distributions for diesel and gasoline engine exhaust particulate matter. Journal of Aerosol Science 32 (6), 749–764.

Harrison, R.M., Deacon, A.R., Jones, M.R., 1997. Sources and processes affecting concentrations of PM10 and PM2.5 particulate matter in Birmingham (U.K.). Atmospheric Environment 31, 4103–4117.

Harrison, R.M., Shi, J.P., Xi, S., Khan, A., Mark, D., Kinnersley, R., Yin, J., 2000. Measurement of number, mass and size distribution of particles in the atmosphere. Philosophical Transactions of the Royal Society of London A 358, 2567–2580.

Harrison, R.M., Tilling, R., Romero, M.S.C., Harrad, S., Jarvis, K., 2003. A study of trace metals and polycyclic aromatic hydrocarbons in the roadside environment. Atmospheric Environment 37, 2391–2402.

Harrison, R.M., Jones, A.M., Barrowcliffe, R., 2004. Field study of the influence of meteorological factors and traffic volumes upon suspended particle mass at urban roadside sites of different geometry. Atmospheric Environment 38, 6361–6369.

Harrison, R.M., 2009. Airborne particulate matter from road traffic: current status of knowledge and research challenges paper presented at the ETTAO09. In: 17th Transport and Air Pollution Symposium – 3rd Environment and Transport Symposium.

Harrison, R.M., Giorio, C., Beddows, D.C., Dall'Osto, M., 2010. Size distribution of airborne particles controls outcomes of epidemiological studies. Science of the Total Environment 409, 289–293.

Harrison, R.M., Beddows, D.CS., Dall'Osto, M., 2011. PMF analysis of wide-range particle size spectra collected on a major highway. Environmental Science and Technology 45, 5522–5528.

Harrison, R.M., Dall'Osto, M., Beddows, D.C.S., Thorpe, A.J., Bloss, W.J., Allan, J.D., Coe, H., Dorsey, J.R., Gallagher, M., Martin, C., Whitehead, J., Williams, P.I., Jones, R.L., Langridge, J.M., Benton, A.K., Ball, S.M., Langford, B., Hewitt, C.N., Davison, B., Martin, D., Petersson, K., Henshaw, S.J., White, I.R., Shallcross, D.E., Barlow, J.F., Dunbar, T., Davies, F., Nemitz, E., Phillips, G.J., Helfter, C., Di Marco, C.F., Smith, S., 2012a. Atmospheric chemistry and physics in the atmosphere of a developed megacity (London): an overview of the REPARTEE experiment and its conclusions. Atmospheric Chemistry and Physics 12, 3065–3114.

Harrison, R.M., Jones, A.M., Gietl, J., Yin, J., Green, D.C., 2012b. Estimation of the contributions of brake dust, tire wear, and resuspension to nonexhaust traffic particles derived from atmospheric measurements. Environmental Science and Technology 46, 6523–6529.

Hassanien, M.A., Abdel-Latif, N.M., 2008. Polycyclic aromatic hydrocarbons in road dust over Greater Cairo, Egypt. Journal of Hazardous Materials 151, 247–254.

Hays, M.D., Cho, S.-H., Baldauf, R., Schauer, J.J., Shafer, M., 2011. Particle size distributions of metal and non-metal elements in an urban near-highway environment. Atmospheric Environment 45 (4), 925–934.

He, L.-Y., Hu, M., Huang, X.-F., Zhang, Y.-H., Yu, B.-D., Liu, D.-Q., 2006. Chemical characterization of fine particles from on-road vehicles in the Wutong tunnel in Shenzhen, China. Chemosphere 62 (12), 1565–1573.

He, L.-Y., Hu, M., Zhang, Y.-H., Huang, X.-F., Yao, T.-T., 2008. Fine particle emissions from on-road vehicles in the Zhujiang tunnel, China. Environmental Science and Technology 42 (12), 4461–4466.

HEI Panel on the Health Effects of Traffic-Related Air Pollution, 2010. Traffic-related Air Pollution: A Critical Review of the Literature on Emissions, Exposure, and Health Effects. HEI Special Report, 17. Health Effects Institute, Boston, MA.

Hien, T.T., Thanh, L.T., Kameda, T., Takenaka, N., Bandow, H., 2007. Distribution characteristics of polycyclic aromatic hydrocarbons with particle size in urban aerosols at the roadside in Ho Chi Minh City, Vietnam. Atmospheric Environment 41 (8), 1575–1586.

Hildemann, L.M., Markowski, G.R., Jones, M.C., Cass, G.R., 1991. Submicrometer aerosol mass distributions of emissions from boilers, fireplaces, automobiles, diesel trucks and meat cooking operations. Aerosol Science and Technology 14 (1), 138–152.

Hjortenkrans, D.S.T., Bergbäck, B.G., Häggerud, A.V., 2007. Metal emissions from brake linings and tires: case studies of Stockholm, Sweden 1995/1998 and 2005. Environmental Science and Technology 41 (15), 5224–5230.

Hopke, P.K., 2003. Recent developments in receptor modelling. Journal of Chemometrics 17, 255–265.

Huang, X.-F., Chen, D.-L., Lan, Z.-J., Feng, N., He, L.-Y., Yu, G.-H., Luan, S.-J., 2012. Characterization of organic aerosol in fine particles in a mega-city of South China: molecular composition, seasonal variation, and size distribution. Atmospheric Research 114–115, 28–37.

Hussein, T., Johansson, C., Karlsson, H., Hansson, H.-C., 2008. Factors affecting non-tailpipe aerosol particle emissions from paved roads: on-road measurements in Stockholm, Sweden. Atmospheric Environment 42 (4), 688–702.

Iijima, A., Sato, K., Yano, K., Tago, H., Kato, M., Kimura, H., Furuta, N., 2007. Particle size and composition distribution analysis of automotive brake abrasion dusts for the evaluation of antimony sources of airborne particulate matter. Atmospheric Environment 41 (23), 4908–4919.

Imhof, D., Weingartner, E., Prevot, A.S.H., Ordonez, C., Kurtenbach, R., Weisen, P., Rodler, J., Sturm, P., McCrae, I., Sjodin, A., Baltensperger, U., 2005a. Aerosol and NOx emissions factors and submicron particle size distributions in two road tunnels with different traffic regimes. Atmospheric Chemistry and Physics Discussions 5, 5127–5166.

Imhof, D., Weingartner, E., Ordonez, C., Gehrig, R., Hill, M., Buchmann, B., Baltensperger, U., 2005b. Real-world emission factors of fine and ultrafine aerosol particles for different traffic situations in Switzerland. Environmental Science and Technology 39 (21), 5127–5166.

Ingo, G.M., D'Uffizi, M., Falso, G., Bultrini, G., Padeletti, G., 2004. Thermal and microchemical investigation of automotive brake pad wear residues. Thermochimica Acta 418, 61–68.

Jamriska, M., Morawska, L., Thomas, S., He, C., 2004. Diesel bus emissions measured in a tunnel study. Environmental Science and Technology 38, 6701–6709.

Jayaratne, E.R., Wang, L., Heuff, D., Morawska, L., Ferreiera, L., 2009. Increase in particle number emissions from motor vehicles due to interruption of steady traffic flow. Transportation Research Part D 14, 521–526.

Johansson, C., Norman, M., Burman, L., 2009. Road traffic emission factors for heavy metals. Atmospheric Environment 43 (31), 4681–4688.

Jones, A.M., Harrison, R.M., 2006. Estimation of emission factors of particle number and mass fractions from traffic at a site where mean vehicle speeds vary over short distances. Atmospheric Environment 35, 7125–7137.

Jones, A.M., Yin, J., Harrison, R.M., 2008. The weekday-weekend difference and the estimation of the non-vehicle contributions to the urban increment of airborne particulate matter. Atmospheric Environment 42, 4467–4479.

Kam, W., Liacos, J.W., Schauer, J.J., Delfino, R.J., Sioutas, C., 2012. Size-segregated composition of particulate matter (PM) in major roadways and surface streets. Atmospheric Environment 55, 90–97.

Karanasiou, A., Moreno, T., Amato, F., Lumbreras, J., Narros, A., Borge, R., Tobias, A., Boldo, E., Linares, C., Pey, J., Reche, C., Alastuey, A., Querol, X., 2011. Road dust contribution to PM levels – evaluation of the effectiveness of street washing activities by means of positive matrix factorization. Atmospheric Environment 45 (13), 2193–2201.

Karanasiou, A., Moreno, T., Amato, F., Tobias, A., Boldo, E., Linares, C., Lumbreras, J., Borge, R., Alastuey, A., Querol, X., 2012. Variation of PM2.5 concentrations in relation to street washing activities. Atmospheric Environment 54, 465–469.

Kaunhaniemi, M., Kukkonen, J., Harkonen, J., Nikmo, J., Kangas, L., Omstedt, G., Ketzel, M., Kousa, A., Haakana, M., Karppinen, A., 2011. Evaluation of a road dust suspension model for predicting the concentrations of PM10 in a street canyon. Atmospheric Environment 45, 3646–3654.

Kelly, F.J., Fussell, J.C., 2012. Size, source and chemical composition as determinates of toxicity attributable to ambient particulate matter. Atmospheric Environment 60, 504–526.

Kerminen, V.-M., Pakkanen, T.A., Mäkelä, T., Hillamo, R.E., Sillanpää, M., Rönkkö, T., Virtanen, A., Keskinen, J., Pirjola, L., Hussein, T., Hameri, K., 2007. Development of particle number size distribution near a major road in Helsinki during an episodic inversion situation. Atmospheric Environment 41 (8), 1759–1767.

Ketzel, M.G., Omstedt, G., Johansson, C., During, I., Pohjola, M., Oettl, D., Gidhagen, L., Wahlin, P., Lohmeyer, A., Berkowicz, 2007. Estimation and validation of PM2.5/PM10 exhaust and non-exhaust emission factors for practical street pollution modelling. Atmospheric Environment 41, 9370–9385.

Keuken, M., Denier van der Gon, H., van der Valk, K., 2010. Non-exhaust emissions of PM and the efficiency of emission reduction by road sweeping and washing in the Netherlands. Science of the Total Environment 408 (20), 4591–4599.

Keuken, M., Henzing, B., Zandveld, P., van den Elshout, S., Karl, M., 2012. Dispersion of particle numbers and elemental carbon from road traffic, a harbour and an airstrip in the Netherlands. Atmospheric Environment 54, 320–327.

Kim, J.Y., Lee, J.Y., Kim, Y.P., Lee, S.B., Jin, H.C., Bae, G.N., 2012. Seasonal characteristics of the gaseous and particulate PAHs at a roadside station in Seoul, Korea. Atmospheric Research 116, 142–150.

Kittleson, D.B., Watts, W.F., Johnson, J.P., 2004. Nanoparticle emissions on Minnesota highways. Atmospheric Environment 38, 9–19.

Klems, J.P., Pennington, M.R., Zordan, C.A., Johnston, M.V., 2010. Ultrafine particles near a roadway intersection: origin and apportionment of fast changes in concentration. Environmental Science and Technology 44 (20), 7903–7907.

Kuhlbusch, T.A.J., John, A.C., Quass, U., 2009. Sources and source contributions to fine particles. Biomarkers 14 (1), 23–28.

Kumar, P., Fennell, P., Langley, D., Britter, R., 2008. Pseudo-simultaneous measurements for the vertical variation of coarse, fine and ultrafine particles in an urban street canyon. Atmospheric Environment 42 (18), 4304–4319.

Kumar, P., Pirjola, L., Ketzel, M., Harrison, R.M., 2013. Nanoparticle emissions from 11 non-vehicle exhaust sources – a review. Atmospheric Environment 67, 252–277.

Kumata, H., Yamada, J., Masuda, K., Takada, H., Sato, Y., Sakural, T., Fujiwara, K., 2002. Benzothiazolamines as tire-derived molecular markers: sorptive behaviours in street runoff and application to source apportioning. Environmental Science and Technology 36 (4), 702–708.

Kupiainen, K.J., Tervahattu, H., Räisänen, M., Mäkelä, T., Aurela, M., Hillamo, R., 2005. Size and composition of airborne particles from pavement wear, tires, and traction sanding. Environmental Science and Technology 39 (3), 699–706.

Kwon, E.E., Castaldi, M.J., 2012. Mechanistic understanding of polycyclic aromatic hydrocarbons (PAHs) from the thermal degradation of tires under various oxygen concentration atmospheres. Environmental Science and Technology 46 (23), 12921–12926.

Laidlaw, M.A.S., Zahran, S., Mielke, H.W., Taylor, M.P., Filippelli, G.M., 2012. Resuspension of lead contaminated urban soil as a dominant source of atmospheric lead in Birmingham, Chicago, Detroit and Pittsburgh, USA. Atmospheric Environment 49, 302–310.

Landis, M.S., Lewis, C.W., Stevens, R.K., Keeler, G.J., Dvonch, J.T., Tremblay, R.T., 2007. Ft. McHenry tunnel study: source profiles and mercury emissions from diesel and gasoline powered vehicles. Atmospheric Environment 41 (38), 8711–8724.

Larsen, R.K., Baker, J.E., 2003. Source apportionment of polycyclic aromatic hydrocarbons in the urban atmosphere: a comparison of three methods. Environmental Science and Technology 37 (9), 1873–1881.

Lee, W.-J., Wang, Y.-F., Lin, T.-C., Chen, Y.-Y., Lin, W.-C., Ku, C.-C., Cheng, J.-T., 1995. PAH characteristics in the ambient air of traffic-source. Science of the Total Environment 159, 185–200.

Liacos, J.W., Kam, W., Delfino, R.J., Schauer, J.J., Sioutas, C., 2012. Characterization of organic, metal and trace element PM2.5 species and derivation of freeway-based emission rates in Los Angeles, CA. Science of the Total Environment 435–436, 5283–5293.

Lim, L.H., Harrison, R.M., Harrad, S., 1999. The contribution of traffic to atmospheric concentrations of polyaromatic hydrocarbons. Environmental Science and Technology 33, 3538–3542.

Lim, J., Yu, L.E., Kostetski, Yu., Lim, C., Ryu, J., Kim, J., 2008. Effects of driving conditions on diesel exhaust particles. Journal of Air and Waste Management Association 58, 1077–1085.

Lin, C.-C., Chen, S.-J., Huang, K.-L., Hwang, W.-I., Chang-Chien, G.-P., Lin, W.-Y., 2005. Characteristics of metals in nano/ultrafine/fine/coarse particles collected beside a heavily Trafficked road. Environmental Science and Technology 39 (21), 8113–8122.

Lin, H.-Y., Hung, C.-H., Yuan, C.-S., Chen, C.-J., Chen, W.-C., Chiang, S.-W., Wang, C.J.-L., 2008. Characteristics and source identification of roadside suspended particles. Transportation Research Part D: Transport and Environment 13 (8), 531–538.

Lingard, J.J.N., Agus, E.L., Young, D.T., Andrews, G.E., Tomlin, A.S., 2006. Observations of urban airborne particle number concentrations during rush-hour conditions: analysis of the number based size distributions and modal parameters. Journal of Environmental Monitoring 8 (12), 1203–1218.

Liu, M., Cheng, S.B., Ou, D.N., Hou, L.J., Gao, L., Wang, L.L., Xie, Y.S., Yang, Y., Xu, S.Y., 2007. Characterization, identification of road dust PAHs in central Shanghai areas, China. Atmospheric Environment 41, 8785–8795.

Lough, G.C., Schauer, J.J., Park, J.-S., Shafer, M.M., DeMinter, J.T., Weinstein, J.P., 2005. Emissions of metals associated with motor vehicle roadways. Environmental Science and Technology 39 (3), 826–836.

Lough, G.C., Christensen, C.G., Schauer, J.J., Tortorelli, J., Mani, E., Lawson, D.R., Clark, N.N., Gabela, P.A., 2007. Development of molecular marker source profiles for emissions from on-road gasoline and diesel vehicle fleets. Journal of the Air and Waste Management Association 57 (10), 1190–1199.

Luo, X.S., Yu, S., Zhu, Y.G., Li, X.D., 2011. Trace metal contamination in urban soils of China. Science of the Total Environment 421–422, 17–30.

Majumdar, D., Rajaram, B., Meshram, S., Chalapati Rao, C.V., 2012. PAHs in road dust: ubiquity, fate, and summary of available data. Critical Reviews in Environmental Science and Technology 42 (12), 1191–1232.

Mancilla, Y., Mendoza, A., 2012. A tunnel study to characterize PM2.5 emissions from gasoline-powered vehicles in Monterrey, Mexico. Atmospheric Environment 59, 449–460.

Manno, E., Varrica, D., Dongarrà, G., 2006. Metal distribution in road dust samples collected in an urban area close to a petrochemical plant at Gela, Sicily. Atmospheric Environment 40 (30), 5929–5941.

Mari, M., Harrison, R.M., Schuhmacher, M., Domingo, J.L., Pongpiachan, S., 2010. Inferences over the sources and processes affecting polycyclic aromatic hydrocarbons in the atmosphere derived from measured data. Science of the Total Environment 408, 2387–2393.

Martuzevicius, D., Kliucininkas, L., Prasauskas, T., Krugly, E., Kauneliene, V., Strandberg, B., 2011. Resuspension of particulate matter and PAHs from street dust. Atmospheric Environment 45 (2), 310–317.

Mathissen, M., Scheer, V., Vogt, R., Benter, T., 2011. Investigation on the potential generation of ultrafine particles from the tire–road interface. Atmospheric Environment 45 (34), 6172–6179.

Maricq, M.M., 2007. Chemical characterization of particulate emissions from diesel engines: a review. Journal of Aerosol Science 38 (11), 1079–1118.

Masiol, M., Hofer, A., Squizzato, S., Piazza, R., Rampazzo, G., Pavoni, B., 2012. Carcinogenic and mutagenic risk associated to airborne particle-phase polycyclic aromatic hydrocarbons: a source apportionment. Atmospheric Environment 60, 375–382.

Mauderly, J.L., 1994. Toxicological and epidemiological evidence for health risks from inhaled engine emissions. Environmental Health Perspectives 102, 165–171.

McLaren, R., Gertler, A.W., Wittorff, D.N., Belzer, W., Dann, T., Singleton, D.L., 1996. Real-world measurements of exhaust and evaporative emissions in the cassiar tunnel predicted by chemical mass balance modeling. Environmental Science and Technology 30, 3001–3009.

Miguel, A.H., Kirchstetter, T.W., Harley, R.A., Hering, S.V., 1998. On-road emissions of particulate polycyclic aromatic hydrocarbons and black carbon from gasoline and diesel vehicles. Environmental Science and Technology 32 (4), 450–455.

Milani, M., Pucillo, F.P., Ballerini, M., Camatani, M., Gualtieri, M., Martino, S., 2004. First evidence of tyre debris characterization at the nanoscale by focused ion beam. Materials Characterization 52, 283–288.

Morawska, L., Jayaratne, E.R., Mengersen, K., Jamriska, M., Thomas, S., 2002. Differences in airborne particle and gaseous concentrations in urban air between weekdays and weekends. Atmospheric Environment 36, 4375–4383.

Nanzetta, K.M., Holmen, B.A., 2004. Roadside particle number distributions and relationships between number concentrations, meteorology, and traffic along a Northern California freeway. Journal of Air and Waste Management Association 54, 540–554.

Narvaez, R.F., Hoepner, L., Chillrud, S.N., Yan, B., Garfinkel, R., Whyatt, R., Camann, D., Perera, F.P., Kinney, P.L., Miller, R.L., 2008. Spatial and temporal trends of polycyclic aromatic hydrocarbons and other traffic-related airborne pollutants in new York city. Environmental Science and Technology 42, 7330–7335.

Nicholson, K.W., 1988. A review of particle resuspension. Atmospheric Environment 22 (12), 2639–2651 (1967).

Nielsen, T., 1996. Traffic contribution of polycyclic aromatic hydrocarbons in the centre of a large city. Atmospheric Environment 30 (20), 3481–3490.

Ntziachristos, L., Ning, Z., Geller, M.D., Sheesley, R.J., Schauer, J.J., Sioutas, C., 2007. Fine, ultrafine and nanoparticle trace element compositions near a major freeway with a high heavy-duty diesel fraction. Atmospheric Environment 41, 5864–5866.

Oliveira, C., Pio, C., Caseiro, A., Santos, P., Nunes, T., Mao, H., Luahana, L., Sokhi, R.S., 2010. Road traffic impact on urban atmospheric aerosol loading at Oporto, Portugal. Atmospheric Environment 44 (26), 3147–3158.

Oliveira, C., Martins, N., Tavares, J., Pio, C., Cerqueira, M., Matos, M., Silva, H., Oliviera, S., Camoes, F., 2011. Size distribution of polycyclic aromatic hydrocarbons in a roadway tunnel in Lisbon, Portugal. Chemosphere 83 (11), 1588–1596.

Omar, N.Y.M.J., Abas, M.R.B., Rahman, N.A., Tahir, N.M., Rushdi, A.I., Simoneit, B.R.T., 2007. Levels and distributions of organic source tracers in air and roadside dust particles of Kuala Lumpur, Malaysia. Environmental Geology 52, 1485–1500.

Omstedt, G., Bringfelt, B., Johansson, C., 2005. A model for vehicle-induced nontailpipe emissions of particles along Swedish roads. Atmospheric Environment 39, 6088–6097.

Oanh, N.T.K., Thiansathit, W., Bond, T.C., Subramanian, R., Winijkul, E., Pawarmart, I., 2010. Compositional characterization of PM2.5 emitted from in-use diesel vehicles. Atmospheric Environment 44 (1), 15–22.

Ondráček, J., Schwarz, J., Žďímal, V., Andělová, L., Vodička, P., Bízek, V., Tsai, C.-J., Chen, S.-C., Smolík, J., 2011. Contribution of the road traffic to air pollution in the Prague city (busy speedway and suburban crossroads). Atmospheric Environment 45 (29), 5090–5100.

Padro-Martinez, L.T., Patton, A.P., Trull, J.B., Zamore, W., Brugge, D., Durant, J.L., 2012. Mobile monitoring of particle number concentration and other traffic-related air pollutants in a near-highway neighbourhood over the course of a year. Atmospheric Environment 61, 253–264.

Pant, P., Harrison, R.M., 2012. Critical review of receptor modelling for particulate matter: a case study of India. Atmospheric Environment 49, 1–12.

Peltier, R.E., Cromar, K.R., Ma, Y., Fan, Z.-H., Lippmann, M., 2011. Spatial and seasonal distribution of aerosol chemical components in New York City: (2) road dust and other tracers of traffic-generated air pollution. Journal of Exposure Science and Environmental Epidemiology 21 (5), 484–494.

Perez, N., Pey, J., Cusack, M., Reche, C., Querol, X., Alastuey, A., Viana, M., 2010. Variability of particle number, black carbon and PM10, PM2.5 and PM1 levels and speciation: Influence of road traffic emissions on urban air quality. Aerosol Science and Technology 44 (7), 487–499.

Perrone, M.G., Larsen, B.R., Ferrero, L., Sangiorgi, G., De Gennaro, G., Udisti, R., Zangranado, R., Gambaro, A., Bolzacchini, E., 2012. Sources of high PM2.5 concentrations in Milan, Northern Italy: molecular marker data and CMB modelling. Science of the Total Environment 414, 343–355.

Pey, J., Querol, X., Alastuey, A., Rodriguez, R., Putuad, J.-P., Van Dingenen, R., 2009. Source apportionment of urban fine and ultra-fine particle number concentration in a Western Mediterranean city. Atmospheric Environment 43, 4407–4415.

Pey, J., Querol, X., Alastuey, A., 2010. Discriminating the regional and urban contributions in the North-Western Mediterranean: PM levels and composition. Atmospheric Environment 44 (13), 1587–1596.

Phuleria, H.C., Geller, M.D., Fine, P.M., Sioutas, C., 2006. Size-resolved emissions of organic tracers from light- and heavy-duty vehicles measured in a California roadway tunnel. Environmental Science and Technology 40 (13), 4109–4118.

Phuleria, H.C., Sheesley, R.J., Schauer, J.J., Fine, P.M., Sioutas, C., 2007. Roadside measurements of size-segregated particulate organic compounds near gasoline and diesel-dominated freeways in Los Angeles, CA. Atmospheric Environment 41 (22), 4653–4671.

Prichard, H.M., Fisher, P.C., 2012. Identification of platinum and palladium particles emitted from vehicles and dispersed into the surface environment. Environmental Science and Technology 46 (6), 3149–3154.

Pulles, T., Denier van der Gon, H., Appelman, W., Verheul, M., 2012. Emission factors for heavy metals from diesel and petrol used in European vehicles. Atmospheric Environment 61, 641–651.

Puustinen, A., Hämeri, K., Pekkanen, J., Kulmala, M., de Hartog, J., Meliefste, K., ten Brink, H., Kos, G., Katsouyanni, K., Karakatsani, A., Kotronarou, A., Kavouras, I., Meddings, C., Thomas, S., Harrison, R., Ayres, J.G., van der Zee, S., Hok, G., 2007. Spatial variation of particle number and mass over four European cities. Atmospheric Environment 41 (31), 6622–6636.

Reche, C., Querol, X., Alastuey, A., Viana, M., Pey, J., Moreno, T., Rodriguez, S., Gonzalez, Y., Fernandez-Camacho, R., Sanchez de la Campa, A.M., de la Rosa, J., Dall'Osto, M., Prevot, A.S.H., Hueglin, C., Harrison, R.M., Quincey, P., 2011. New considerations for PM, black carbon and particle number concentration for air quality monitoring across different European cities. Atmospheric Chemistry and Physics 11, 6207–6227.

Reddy, C.M., Quinn, J.G., 1997. Environmental chemistry of benzothiazoles derived from rubber. Environmental Science and Technology 31, 2847–2853.

Rexeis, M., Hausberger, S., 2009. Tend of vehicle emission levels until 2020-Prognosis based on current vehicle measurements and future emission legislation. Atmospheric Environment 43, 4689–4698.

Riddle, S.G., Robert, M.A., Jakober, C.A., Hannigan, M.P., Kleeman, M.J., 2007. Size distribution of trace organic species emitted from light-duty gasoline vehicles. Environmental Science and Technology 41 (21), 7464–7471.

Rissler, J., Swietlicki, E., Bengtsson, A., Boman, C., Pagels, J., Sandstrom, T., Blomberg, A., Londahl, J., 2012. Experimental determination of deposition of diesel exhaust particles in the human respiratory tract. Journal of Aerosol Science 48, 18–33.

Robert, M.A., VanBergen, S., Kleeman, M.J., Jakober, C.A., 2007a. Size and composition distributions of particulate matter emissions: part 1 – light-duty diesel vehicles. Journal of Air and Waste Management Association 57, 1414–1428.

Robert, M.A., Kleeman, M.J., Jakober, C.A., 2007b. Size and composition distributions of particulate matter emissions: part 2 – heavy-duty diesel vehicles. Journal of Air and Waste Management Association 57, 1429–1438.

Robinson, A.L., Subramanian, R., Donahue, N.M., Rogge, W.F., 2006. Source apportionment of molecular markers and organic Aerosol1. Polycyclic aromatic hydrocarbons and methodology for data visualization. Environmental Science and Technology 40 (24), 7803–7810.

Rodriguez, S., Van Dingenen, R., Putuad, J.-P., Dell'Acqua, A., Pey, J., Querol, X., Alastuey, A., Chenery, S., Ho, K.-F., Harrison, R., Tardivo, R., Scarnato, B., Gemelli, V., 2007. A study on the relationship between mass concentrations, chemistry and number size distribution of urban fine aerosols in Milan, Barcelona and London. Atmospheric Chemistry and Physics 35, 2217–2232.

Rogak, S.N., Green, S.I., Pott, U., 1998. Use of tracer gas for direct calibration of emission-factor measurements in a traffic tunnel. Journal of the Air and Waste Management Association 48 (6), 545–552.

Rogge, W.F., Hildemann, L.M., Mazurek, M.A., Cass, G.R., Simoneit, B.R.T., 1993. Sources of fine organic aerosol. 3. Road dust, tire debris, and organometallic brake lining dust: roads as sources and sinks. Environmental Science and Technology 27 (9), 1892–1904.

Rose, D., Wehner, B., Ketzel, M., Engler, C., Voigtlander, J., Tuch, T., Wiedensohler, A., 2006. Atmospheric number size distributions of soot particles and estimation of emission factors. Atmospheric Chemistry and Physics 6 (4), 1021–1031.

Rosenbohm, E., Vogt, R., Scheer, V., Nielsen, O.J., Dreiseidler, A., Baumbach, G., Imhof, D., Baltensperger, U., Fuchs, J., Jaeschke, V., 2006. Particulate size distributions and mass measured at a motorway during the BAB II campaign. Atmospheric Environment 37, 5283–5293.

Ruellan, S., Cachier, H., 2001. Characterisation of fresh particulate vehicular exhausts near a Paris high flow road. Atmospheric Environment 35 (2), 453–468.

Rutter, A.P., Snyder, D.C., Schauer, J.J., Sheesley, R.J., Olson, M.R., DeMinter, J., 2011. Contributions of resuspended soil and road dust to organic carbon in fine particulate matter in the Midwestern US. Atmospheric Environment 45 (2), 514–518.

Sadiktsis, I., Bergvall, C., Johansson, C., Westerholm, R., 2012. Automobile Tires—a potential source of highly carcinogenic dibenzopyrenes to the environment. Environmental Science and Technology 46 (6), 3326–3334.

Sahu, M., Hu, S., Ryan, P.H., Le Masters, G., Grinshpun, S.A., Chow, J.C., Biswas, P., 2011. Chemical compositions and source identification of PM2.5 aerosols for estimation of a diesel source surrogate. Science of the Total Environment 409 (13), 2642–2651.

Samara, C., Voutsa, D., 2005. Size distribution of airborne particulate matter and associated heavy metals in the roadside environment. Chemosphere 59, 1197–1206.

Sanchez-Ccoyllo, O.R., Ynoue, R.Y., Martins, L.D., Astolfo, R., Miranda, R.M., Freitas, E.D., Borges, A.S., Fornaro, A., Freitas, H., Moreira, A., Andrade, M.F., 2009. Vehicular particulate matter emissions in road tunnels in Sao Paulo, Brazil. Environmental Monitoring and Assessment 149 (1–4), 241–249.

Sanders, P.G., Xu, N., Dalka, T.M., Maricq, M.M., 2003. Airborne brake wear debris: size distribution, composition, and a comparison of dynamometer and vehicle tests. Environmental Science and Technology 37 (4), 4060–4069.

Schaap, M., Manders, A.M.M., Hendriks, E.C.J., Cnossen, J.M., Segers, A.J.S., Denier van der Gon, H.A.C., Jozwicka, M., Sauter, F., Velders, G., Matthijsen, J., Builtjes, P.J.H., 2009. Regional Modelling of PM10 over the Netherlands. Technical Report 500099008. Available at: http://www.rivm.nl/bibliotheek/rapporten/500099008.pdf (accessed 09.09.12).

Schauer, J.J., Rogge, W.F., Hildemann, L.M., Mazurek, M.A., Cass, G.R., Simoneit, B.R.T., 1996. Source apportionment of airborne particulate matter using organic compounds as tracers. Atmospheric Environment 30, 3837–3855.

Schauer, J.J., Lough, G.C., Shafer, M.M., Christensen, W.C., Arndt, M.F., DeMinter, J.T., Park, J.-S., 2006. Characterization of Emissions of Metals Emitted from Motor Vehicles. Health Effects Institute.

Shen, Z., Han, Y., Cao, J., Tian, J., Zhu, C., Liu, S., Liu, P., Wang, Y., 2010. Characteristics of traffic-related emissions: a case study in roadside ambient air over Xi'an, China. Aerosol and Air Quality Research 10, 292–300.

Shi, J.P., Harrison, R.M., 1999. Investigation of ultrafine particle formation during diesel exhaust dilution. Environmental Science and Technology 33, 3730–3736.

Shi, J.P., Khan, A.A., Harrison, R.M., 1999. Measurements of ultrafine particle concentration and size distribution in the urban atmosphere. Science of the Total Environment 235, 51–64.

Sjodin, A., Ferm, M., Bjork, A., Rahmberg, M., Gudmundsson, A., Swietlickli, E., Johansson, C., Gustafsson, M., Blomqvist, G., 2010. Wear Particles from Road Traffic: A Field, Laboratory and Modelling Study. IVL Report B1830. Available at: http://www.ivl.se/webdav/files/B-rapporter/B1830.pdf.

Smith, D.J.T., Harrison, R.M., 1996. Concentrations, trends and vehicle source profile of polynuclear aromatic hydrocarbons in the U.K. atmosphere. Atmospheric Environment 30 (14), 2513–2525.

Sodeman, D.A., Toner, S.M., Prather, K.A., 2005. Determination of single particle mass spectral signatures from light-duty vehicle emissions. Environmental Science and Technology 39 (12), 4569–4580.

Song, F., Gao, Y., 2011. Size distributions of trace elements associated with ambient particular matter in the affinity of a major highway in the New Jersey–New York metropolitan area. Atmospheric Environment 45 (37), 6714–6723.

Song, S., Wu, Y., Jiang, J., Yang, L., Cheng, Y., Hao, J., 2012. Chemical characteristics of size-resolved PM(2.5) at a roadside environment in Beijing, China. Environmental Pollution 161, 215–221.

Sonntag, D.B., Bailey, C.R., Fulper, C.R., Baldauf, R.W., 2012. Contribution of lubricating oil to particulate matter emissions from light-duty gasoline vehicles in Kansas city. Environmental Science and Technology 46 (7), 4191–4199.

Stechmann, H., Dannecker, W., 1990. Characterization and source analysis of vehicle-generated aerosols. Journal of Aerosol Science 21 (Suppl. 1), S287–S290.

Staehelin, J., Schlapfer, K., Burgin, T., Steinemann, U., Schneider, S., Brunner, D., Baumle, M., Meier, M., Zahner, C., Keiser, S., Stahel, W., Keller, C., 1995. Emission factors from road traffic from a tunnel study (Gubrist tunnel, Switzerland). Part I: concept and first results. Science of the Total Environment 169, 141–147.

Sternbeck, J., Sjödin, Å., Andréasson, K., 2002. Metal emissions from road traffic and the influence of resuspension—results from two tunnel studies. Atmospheric Environment 36 (30), 4735–4744.

Sun, Y.L., Zhang, Q., Schwab, J.J., Chen, W.-N., Bae, M.-S., Hung, H.-M., Lin, Y.-C., Ng, N.L., Jayne, J., Massoli, P., Williams, L.R., Demerjian, K.L., 2012. Characterization of near-highway submicron aerosols in New York City with a high-resolution aerosol mass spectrometer. Atmospheric Chemistry and Physics 12, 2215–2227.

Sysalova, J., Sykorova, I., Havelcova, M., Szakova, J., Trejtnarova, H., Kotlik, B., 2012. Toxicologically important trace elements and organic compounds investigated in size –fractionated urban particulate matter collected near the Prague highway. Science of the Total Environment 437, 127–136.

Tanner, P.A., Hoi-Ling, M., Yu, P.K.N., 2008. Fingerprinting metals in street dust in Beijing, Shanghai and Hong Kong. Environmental Science and Technology 42, 7111–7117.

Teixiera, E.C., Agudelo-Castaneda, D.M., Fachel, J.M.G., Leal, K.L., Garcia, K.O., Wiegand, F., 2012. Source identification and seasonal variation of polycyclic aromatic hydrocarbons associated with atmospheric fine and coarse particles in the Metropolitan Area of Porto Alegre, RS, Brazil. Atmospheric Research 118, 390–403.

Tervahattu, H., Kupiainen, K.J., Raisanen, M., Makela, T., Hillamo, R., 2006. Generation of urban road dust from anti-skid and asphalt concrete aggregates. Journal of Hazardous Materials 132, 39–46.

Thorpe, A.J., Harrison, R.M., Boulter, P.G., McCrae, I.S., 2007. Estimation of particle resuspension source strength on a major London Road. Atmospheric Environment 41, 8007–8020.

Thorpe, A., Harrison, R.M., 2008. Sources and properties of non-exhaust particulate matter from road traffic: a review. Science of the Total Environment 400 (1–3), 270–282.

Van Poppel, M., Panis, L.I., Govarts, E., Van Houtte, J., Maenhaut, W., 2012. A comparative study of traffic related air pollution next to a motorway and a motorway flyover. Atmospheric Environment 60, 132–141.

Varrica, D., Bardelli, F., Dongarra, G., Tamburro, E., 2012. Speciation of Sb in airborne particulate matter, vehicle brake linings and brake pad wear residues. Atmospheric Environment 64, 18–24.

Viana, M., Kuhlbusch, T.A.J., Querol, X., Alastuey, A., Harrison, R.M., Hopke, P.K., Winiwarter, W., Vallius, M., Szidat, S., Prevot, A.S.H., Hueglin, C., Bloemen, H., Wahlin, P., Vecchi, R., Miranda, A.I., Kasper-Giebl, A., Maenhaut, W., Hitzenberger, R., 2008. Source apportionment of particulate matter in Europe: a review of methods and results. Aerosol Science 39, 827–849.

Wahlin, P., Berkowicz, R., Palmgren, F., 2006. Characterisation of traffic-generated particulate matter in Copenhagen. Atmospheric Environment 40 (12), 2151–2159.

Wang, G., Kawamura, K., Xie, M., Hu, S., Gao, S., Cao, J., An, Z., Wang, Z., 2009. Size distribution on n-alkanes, PAHs and hopanes and their sources in the urban, mountain and marine atmospheres over East Asia. Atmospheric Chemistry and Physics 14, 8869–8882.

Wang, F., Ketzel, M., Ellermann, T., Wahlin, P., Jensen, S.S., Fang, D., Massling, A., 2010. Particle number, particle mass and NOx emission factors at a highway and an urban street in Copenhagen. Atmospheric Chemistry and Physics 10, 2745–2764.

Wang, X., Westerdahl, D., Wu, Y., Pan, X., Zhang, K.M., 2011. On-road emission factor distributions of individual diesel vehicles in and around Beijing, China. Atmospheric Environment 45 (2), 503–513.

Watson, J.G., Chow, J.C., Lowenthal, D.H., Pritchett, L.C., Frazier, C.A., Neuroth, G.R., Robbins, R., 1994. Differences in the carbon composition of source profiles for diesel- and gasoline-powered vehicles. Atmospheric Environment 28 (15), 2493–2505.

Watson, J.G., Zhu, T., Chow, J.C., Engelbrecht, J., Fujita, E.M., Wilson, W.E., 2002. Receptor modeling application framework for particle source apportionment. Chemosphere 49 (9), 1093–1136.

Watson, J.G., Chow, J.C., 2007. Receptor models for source apportionment of suspended particles. In: Murphy, B., Morrison, R. (Eds.), Introduction to Environmental Forensics, second ed., vol. 2. Academic Press, New York, NY, pp. 279–316.

Wehner, B., Birmili, W., Gnauk, T., Wiedensohler, A., 2002. Particle number size distributions in a street canyon and their transformation into the urban-air background: measurements and a simple model study. Atmospheric Environment 36, 2215–2223.

Wei, B., Yang, L., 2010. A review of heavy metal contaminations in urban soils, urban road dusts and agricultural soils from China. Microchemical Journal 94 (2), 99–107.

Weingartner, E., Keller, C., Stahel, W.A., Burtscher, H., Baltensperger, U., 1997. Aerosol emission in a road tunnel. Atmospheric Environment 31 (3), 451–462.

Weijers, E.P., Khlystov, A.Y., Kos, G.P.A., Erisman, J.W., 2004. Variability of particulate matter concentrations along roads and motorways determined by a moving measurement unit. Atmospheric Environment 38, 2993–3002.

Westerdahl, D., Wang, X., Pan, X., Zhang, M.K., 2009. Characterization of on-road vehicle emission factors and microenvironmental air quality in Beijing, China. Atmospheric Environment 43, 697–705.

Wik, A., Dave, G., 2009. Occurrence and effects of tire wear particles in the environment – a critical review and an initial risk assessment. Environmental Pollution 157 (1), 1–11.

Wu, S.-P., Tao, S., Zhang, Z.-H., Lan, T., Zuo, Q., 2007. Characterization of TSP-bound n-alkanes and polycyclic aromatic hydrocarbons at rural and urban sites of Tianjin, China. Environmental Pollution 147 (1), 203–210.

Yan, B., Zheng, M., Hu, Y., Ding, X., Sullivan, A.P., Weber, R.J., Baek, J., Edgerton, E.S., Russell, A.G., 2009. Roadside, urban, and rural comparison of primary and secondary organic molecular markers in ambient PM2.5. Environmental Science and Technology 43 (12), 4287–4293.

Yin, J., Harrison, R.M., Chen, Q., Rutter, A., Schauer, J.J., 2010. Source apportionment of fine particles at urban background and rural sites in the UK atmosphere. Atmospheric Environment 44, 841–851.

Zhang, Y., Stedman, D.H., Bishop, G.A., Guenther, P.L., Beaton, S.P., 1995. Worldwide on-road vehicle exhaust emissions study by remote sensing. Environmental Science and Technology 29, 2286–2294.

Zhu, Y., Hinds, W.C., Kim, S., Sioutas, C., 2002a. Concentration and size distribution of ultrafine particles near a major highway. Journal and Air and Waste Management Association 52, 1032–1042.

Zhu, Y., Hinds, W.C., Kim, S., Shen, S., Sioutas, C., 2002b. Study of ultrafine particles near a major highway with heavy-duty diesel traffic. Atmospheric Environment 36 (27), 4323–4335.

Zhu, Y., Kuhn, T., Mayo, P., Hinds, W.C., 2006. Comparison of daytime and nighttime concentration profiles and size distributions of ultrafine particles near a major highway. Environmental Science and Technology 40, 2531–2536.

Zhu, C.-S., Chen, C.-C., Cao, J.-J., Tsai, C.-J., Chou, C.C.-K., Liu, S.-C., Roam, G.-D., 2010. Characterization of carbon fractions for atmospheric fine particles and nanoparticles in a highway tunnel. Atmospheric Environment 44, 2668–2673.