Assessment of population exposure to air pollution during commuting in European cities

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Front page picture:
Intensive commuting at the busy Avenida Diagonal in Barcelona near the metro and tram station Palau Reial.

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1 Introduction

The association between traffic-related air pollution and health is becoming well established from both epidemiological and toxicological studies (WHO 2005 and 2013). National authorities and the European Union have developed policies and implemented legislation to limit and reduce exposure to ambient air pollutants. However, these mitigation strategies are currently not as effective as expected to abate particulate matter (PM levels, nitrogen dioxide NO_2 and ozone O_3) (Heal et al., 2012). In addition to decreasing emissions and keep on the efforts to decrease concentrations, one potential solution would be to reduce population exposure by managing the actual exposure.

Commuting is considered as one of the high-exposure periods among various daily activities, especially in high vehicle-density metropolitan areas (Duci et al., 2003). Many studies confirm that commuting accounts for high contributions in total personal exposure. Indeed, during their regular journeys commuters can receive up to 30% of their inhaled daily dose of black carbon (BC), and approximately 12% of their daily $PM_{2.5}$ personal exposure, even though such individuals usually travel for no more than 6% of the day (Dons et al. 2011 and 2012; Fondelli et al., 2008).

There is a growing awareness of the need to change our transportation habits by reducing our use of cars and shifting instead to active transport, i.e. walking or cycling. Such change can bring about significant benefits for our health and environment. Having a more active way of transport has beneficial health effects due to increase physical activity for the person in question (de Hartog et al, 2010; Rojas-Rueda et al, 2012). A modal shift towards active transport will also lead to lower emissions and concentrations. Although potential improvements in air quality might be small it will be beneficial for the whole urban population. To help policy makers, urban planners and local administrators make the appropriate choices, it is necessary to quantify all the significant impacts of such a shift. Health impacts of the physical activity and of air pollution reduction are especially important.

The primary aim of this paper is to review the studies performed to date in order to better understand population exposure during commuting by different modes of transport, and to suggest potential strategies to minimise exposure. We note that dose assessment (assessing the actual doses received by the commuters), which is a complementary yet distinct concept to that of exposure, is not the focus of this review.

2 Materials and methods

2.1 Study identification and selection

An electronic search on the ISI web of knowledge database and Google Scholar was conducted using various combinations: “air pollutants”, “black carbon”, “elemental carbon”, “ultrafine particle”, “transport mode”, “commuter”, “exposure” “public transport”, “microenvironment”, “vehicle”, “car”, “automobile”, “bus”, “cyclist”, “bicycle”, “underground system”, “metro”, “subway” without restrictions of publication type or publication date. The reference lists of studies identified by this method were reviewed for links to additional literature. In addition recent articles in relevant journals were collected. Only the studies conducted in Europe are included in this paper. We present the results of 50 exposure studies performed across 4 transport modes: car, bicycle, bus, and subway.

2.2 Air pollutants studied

The main air pollutants that have been determined in different commuting environments include:

- **PM mass concentrations** (Zuurbier et al., 2010; Aarnio et al., 2005: Adams et al., 2001; Adams et al., 2002). Alm et al., 1999; Asmi et al., 2009; Berghams et al., 2009; Boogard et al., 2009; Boudet et al., 1998; Branis et al., 2006; Briggs et al., 2008; Colombi et al., 2013; de Nazelle et al., 2012; Diapouli et al., 2008; Fondelli et al., 2008; Gee et al., 1999 Gee and Raper 1999; Geiss et al., 2010; Int Panis et al., 2010; Jacobs et al., 2010; Johansson and Johansonn 2003; McNabola et al., 2008; Molle et al., 2013; Querol et al., 2012; Raut et al., 2009; Ripanucci et al., 2006 Salma et al., 2007; Seaton et al., 2005; Strak et al., 2010; Kingham et al., 1998; Pfeifer et al., 1999, Dennekamp et al., 2002; Rank et al., 2000)
- **Particle number** concentrations (Aarnio et al., 2005;Asmi et al., 2009; Berghmans et al., 2009; Boogard et al., 2009; Diapouli et al., 2008; Kaur et al., 2006; Molle et al., 2013).
- **Carbon monoxide,** (Alm et al., 1999; Bevan et al., 1991; De Bruin et al., 2004; deNazelle et al., 2012, Dor et al., 1995; Kaur et al., 2005; 2007; Vellopoulou and Ashmore 2009<1998?>; Duci et al., 2003; Moullet et al., 1998).
- **Black or elemental carbon** (deNazelle et al., 2012, Dons et al., 2012, 2013; Fromme et al., 1998; Adams et al., 2002, Moullet<Le Moulle??> et al., 1998).
- **Volatile organic compounds** (Barrefors et al., 1996; Bevan et al., 1991; Fromme et al., 1998; Lofgren et al., 1991; McNabola et al., 2008; Parra et al. 2008; Barrefors and Petersson 1996).
- **Nitrogen dioxide (**Molle et al., 2013)

3 Results

3.1 Exposure during cycling

A total of 21 studies have been identified that calculated the exposure during cycling (Dons et al., 2011, 2012 and 2013; DeNazelle et al., 2012; Nwokoro et al., 2012; Rojas-Rueda et al., 2011; Strak et al., 2010; Zuurbier et al., 2010; Int Panis et al., 2010; Berghmans et al., 2009; Boogaard et al., 2009; Briggs et al., 2008; McNabola et al., 2008; O'Donoghue et al., 2007; Kaur et al., 2006; Adams et al., 2001 and 2002; Rank et al., 2000; Gee and Raper 1999; Kingham et al., 1998; van Wijnen et al., 1995; Bevan et al., 1991). The majority of these studies have been performed in the United Kingdom (UK), and the Netherlands.
Naturally, the exposure levels for cyclists depend on the detailed conditions of the daily trip and will therefore vary significantly among the studied areas. A cyclist riding in the middle of a busy road will be exposed to concentrations higher than those at the kerb site. By increasing the distance to the main traffic flow, e.g. on a separate bicycle lane, the exposure might be substantially lower. For Barcelona deNazelle et al. (2012) report for cyclist average exposure to $PM_{2.5}$ of 35 µg/m^3; for a car drivers the averaged exposure is 36 µg/m^3. Kaur et al. (2005) and Adams et al. (2001) observed slightly higher exposure concentrations for cyclists (34 µg/m^3) in London; averaged exposure for car drivers was 38 µg/m^3. In Dublin the exposure of the cyclist depends on the route taken. For two different route average exposures to $PM_{2.5}$ for cyclists were 88 and 72 µg/m^3 while the corresponding values for a car driver were 83 and 89 µg/m^3 (McNabola et al., 2008).

Concerning particle number concentration for the London urban area Kaur et al. (2006) report an average value of 84,000 particles/cm^3 (for a car driver the average was 37,000 particles/cm^3) while in Brussels and in 11 Dutch cities this was both for cyclist and car drivers <30,000 particles/cm^3 (Int Panis et al., 2010; Boogaard et al., 2009).

Exposure to air pollution while cycling in urban areas is generally considered high if taking into account that the minute ventilation (volume of air per minute) of cyclists is 1-5 times the minute ventilation of car and bus passengers (Zuurbier et al., 2009; Int Panis et al., 2010). Quantities of particles inhaled by cyclists can be between 4-7 times higher compared to car passengers on the same route, (Int Panis et al., 2010; O'Donoghue et al., 2007). In London, commuting to work by bicycle has been associated with increased long-term inhaled dose of BC (Nwokoro et al., 2012).

3.2 Exposure in cars

Several studies exist on the personal exposure to air pollutants inside private vehicles (Zuurbier et al., 2010 and 2011; Geiss et al., 2010, Asmi et al., 2009; Cattaneo et al., 2009; McNabola et al., 2008; Alm et al., 1999). The main conclusion from all studies is that commuters' exposure is significantly influenced by the traffic intensity. The air pollutants concentrations inside a private car depend on the ambient air pollutant concentration the choice of ventilation used inside the cars and the type of fuel used (Geiss et al., 2010; Cattaneo et al., 2009; Briggs et al., 2008; Diapouli et al., 2008; Rank et al., 2001). Typical levels for commuter exposure in passenger cars are in the range 35-75 µgm^{-3} for PM_{10} and 22-85 µgm^{-3} for $PM_{2.5.}$ Adams et al. (2001) and Kaur et al. (2005) obtained $PM_{2.5}$ exposure measurements of 35 µg/m^3. However some studies found that pollutants concentrations inside the car depend highly on the type of fuel used (Jalava et al., 2012; Zuurbier et al., 2010).

3.3 Exposure in buses

Exposure levels in buses are quite variable across the different countries. Asmi et al. (2009) found moderate exposure for bus drivers and bus commuters in Helsinki. $PM_{2.5}$ average concentrations inside new buses were 12 µg/m^3 only slightly increased when compared to the background air in Helsinki area (average value of 9 µg/m^3). In new buses (Euro3) concentrations are lower than in Euro2 buses; the passengers are exposed to somewhat higher concentrations than the bus driver (Asmi et al, 2009) due to the driver´s compartment isolation. However, tese low values were expected since the background mass concentrations and also the traffic density in Helsinki are lower than in other European cities like London. Higher $PM_{2.5}$ exposure values (35-37 µg/m^3) were found in London and Aberdeen (Kaur et al., 2005; Dennekamp et al., 2002). Lower $PM_{2.5}$ exposure levels (26 µg/m^3) were observed in Barcelona probably due to the clean bus fleet of the city (deNazelle et al., 2012). McNabola et al. (2008) found average $PM_{2.5}$ exposure of 128 µg/m^3. In the study of Fondelli et al. (2008) conducted in Florence the $PM_{2.5}$ average concentration was 56 µg/m^3. In a recent study conducted in Paris, $PM_{2.5}$ concentration inside buses was 59 µg/m^3 with no significant differences in concentration within the cabin (front, middle, rear) while the outdoor concentration was 30% lower (Molle et al, 2013). Praml and Shierl (2000) found high PM_{10} concentrations inside buses in Munich area with the average value equal to 131 µg/m^3.

3.4 Exposure in underground train systems

Previous studies have found high PM levels in the subway systems of Milan (Colombi et al., 2013), Barcelona (Querol et al., 2012), Berlin (Fromme et al., 1998), Budapest (Salma et al., 2007, 2009), Helsinki (Aarnio et al., 2005), London (Seaton et al., 2005; Adams et al., 2001, Pfeifer et al., 1999), Rome (Ripanucci et al., 2006), Paris (Raut et al., 2009), Prague (Branis, 2006) and Stockholm (Johansson and Johansson, 2003). PM levels are much elevated on the platforms, being around 3-4 times higher than inside trains. The mean PM_{10} levels on the platform ranged from 51 to 407 µg/m^3 while $PM_{2.5}$ levels ranged from 33 to 375 µg/m^3, with the highest concentrations found in Stockholm, Rome and London undergrounds. These variations in PM levels among the metro systems could be explained by the abrasion of railways and catenary metal, and to braking systems (Salma et al., 2007). The lowest $PM_{2.5}$ exposure during metro commuting was obtained in Barcelona in the new constructed lines (27 µg/m^3). Recently, Querol et al. (2012) concluded that the ventilation system and air conditioning inside the trains decisively improve air quality in the subway system. The highest $PM_{2.5}$ exposure has been found in London subway, 202-247 µg/m^3.

3.5 Comparison studies

In table 1 we summarise the PM exposure studies conducted in Europe that compare different commuting modes. It should be mentioned that PM concentrations were measured with a variety of techniques and methods e.g Boogaard et al. (2009) determined $PM_{2.5}$ by the DustTrak that uses light scattering and a factory calibration to express the measurement results in mass units while Adams et al., 2001 used a high flow personal sampler.

Several studies (Boogard et al., 2009; de Nazelle et al., 2012, Zuurbier et al., 2010; Kaur et al., 2005; Adams et al., 2001) report lower exposures to PM for cyclists than for car-passengers. In Netherlands the overall mean concentration of $PM_{2.5}$ during driving was 11% higher than during cycling (Boogaard et al., 2009). However, some studies observed that exposure to fine particles is higher during cycling than during commuting with a private car (McNabola et al., 2008; Gee et al., 1999; Gee and Raper 1999).

Personal exposure to PM for bus passengers depends strongly on the type of bus used and consequently varies significantly among the studied areas. For example in Barcelona where the bus fleet is quite modern (a large proportion is natural gas fuelled) the exposure levels in buses are lower than in cars and when commuting by bicycle (deNazelle et al., 2012). On the other hand in Dublin public bus passengers (diesel-powered) generally display the highest personal exposure to $PM_{2.5}$ compared to other modes (McNabola et al., 2008). Commuting in the subway had the lowest exposure levels in Barcelona (deNazelle et al., 2012 and Querol et al., 2012), whereas mean exposure levels on the London underground rail system were 3-8 times higher than in the other transport modes (Adams et al., 2001). Kaur et al (2005) concluded that $PM_{2.5}$ exposure experience during commuting for each of the transport modes in London are up to three times higher than observed at the fixed measurement stations.

Given the increasing interest in characterising the black carbon BC component of particulate matter, since this is a good marker for the combustion-derived particles and is strongly associated with health outcomes in epidemiological studies (Heal et al., 2012), we also reviewed personal exposure studies reporting on BC levels during different modes of commuting (Table 2). In these studies lower BC exposures are reported for cyclists and higher exposure during commuting in motorised traffic (De Nazelle et al., 2012; Adams et al., 2002; Dons et al., 2012).

Concerning other pollutants such as CO and VOCs, again the car passengers are exposed to higher levels when compared to other transport modes. In Athens the CO exposure over trips of 30 min was significantly higher for private car than for bus and electric train (Duci et al., 2003), reflecting the fact that in urban areas the highest daily exposure to CO occurs during driving (De Bruin et al., 2004; Vellopoulou et al., 1998; Dor et al., 1995). In Berlin a comparison between subway and car exposures showed significantly higher concentrations of PAHs in the subway train (Fromme et al., 1998). McNabola et al. (2008) report that car commuters have the highest exposure to VOCs followed by cyclists and bus passengers. With reference to particle number counts exposures Zuurbier et al. (2010) found that were highest in diesel buses (38,500 particles/cm^3) and for cyclists along a high-traffic intensity route (46,600 particles/cm^3) and lowest in electric buses (29,200 particles/cm^3).

Nevertheless, the inhalation dose of commuters could be different from the exposure levels. Dons et al. (2012) found that exposure to BC during travel in motorized transport is clearly higher than exposure while walking or biking (6.3 μg/m^3 versus 3.4μg/m^3), but when accounting for inhaled doses, this relationship is reversed. Cyclists seem to receive the highest inhaled $PM_{2.5}$ dose followed by bus and car passengers (McNabola et al., 2008). However in the present study we do not examine the inhaled dose of pollutants during commuting due to the lack of studies determining it and also its complexity of interacting factors such as breathing rate, ventilation and/or particle deposition to the respiratory system.

Table1. European studies comparing PM exposure levels (μg/m^3) in different commuting modes

PM exposure	Car	Bus	Bicycle	Taxi	Subway
Barcelona, $PM_{2.5}$ (deNazelle et al., 2012)	36	26	35	–	–
Barcelona, $PM_{2.5}$ (Querol et al., 2012)					27
Dutch cities, $PM_{2.5}$ (Boogard et al., 2009)	14-122	–	6-112	–	–
Dublin, $PM_{2.5}$ (McNabola et al., 2008); route 1	83	128	88	–	–
Dublin, $PM_{2.5}$ (McNabola et al., 2008) ; route 2	89	104	72		
Florence, $PM_{2.5}$ (Fondelli et al., 2008)	–	56	–	39	–
Belgian cities, PM_{10} (Int Panis et al., 2010)	38-74	–	50-73	–	–
London, $PM_{2.5}$ (Kaur et al., 2005)	38	35	34	42	–
Aberdeen, $PM_{2.5}$ (Dennekamp et al., 2002)	11	38	–	–	–
London, $PM_{2.5}$ (Pfeifer et al., 1999)	–	–	–	33	246
London $PM_{2.5}$ (Adams et al., 2001a)	36	39	29	–	202
Manchester, PM_4, (Gee et al., 1999; Gee and Raper, 1999)	42	338	54	–	–
Arnhem, $PM_{2.5}$ (Zurbier et al., 2010)	73-88	60-73	66-71	–	–
Arnhem, PM_{10} (Zurbier et al., 2010)	42-45	57-61	35-37	–	–

Table 2. European studies comparing BC exposure levels (μg/m^3) in different commuting modes

BC exposure	Car	Bus	Bicycle	Taxi	Subway
Barcelona (deNazelle et al., 2012)	17	6	10		
Berlin (Fromme et al., 2008; EC)	8-14				6-109
Antwerpen (Dons et al., 2012)	6	6	4		5
London (Adams et al., 2002)	26-34	16-25	15-19		

3.6 *Other parameters investigated*

Numerous variables influence personal exposure in transport. Kaur et al. (2007) classified potential confounders in four categories: personal factors, mode of transport factors, road traffic factors and meteorological factors. Personal factors (e.g. breathing rates) are not discussed in this paper.

3.6.1 Traffic conditions and route

Traffic emissions and consequently the selected route (low or high traffic) can influence the exposure levels during commuting. It has been proven that in a moving vehicle, the concentration of pollutants depends on the exhaust emissions of neighbouring vehicles (Dor et al., 1995). Indeed, Dons et al. (2012 and 2013) found that on traffic peak hours with high traffic intensities in-car BC concentrations are about 2 $\mu g/m^3$ higher than average. Higher exposures to particle number concentrations were also observed in heavy-traffic areas and during rush hours (Diapouli et al., 2008). Fine particle number, mass concentrations and VOCs exposure inside buses in Northern Spain was found to be related to traffic emissions (Parra et al., 2008). Similarly, in Sweden the exposure of bus passengers to volatile hydrocarbons was related to traffic emissions (Barrefors and Petersson 1996).

Personal exposure for cyclists is substantially reduced by decreasing proximity to motorised traffic, and traffic intensity (Dons et al., 2013; Boogaard et al., 2009; Kaur et al., 2006; Adams et al., 2001b; Kinghams et al., 1998). The average particle number concentration can be 59% higher, while the average soot concentration 39% higher on high-traffic routes than on low-traffic routes. On the contrary exposure to PM_{10} is not significantly affected by the traffic intensity or the distance of the cycle path from the motorised traffic as PM_{10} dependence on primary vehicle emissions is lower than that of particle number and BC concentrations (Strak et al., 2010).

3.6.2 Travel speed

Travel speed is also strongly related to personal exposure. Exposure to CO and fine particles has been found to be considerably higher in the slower than in the faster speeds (Alm et al., 1999). The in-vehicle BC concentrations are elevated at lower speeds (up to 30 km/h) and at speeds above 80 km/h (Dons et al., 2013). The relevance of speed is two-fold: (a) because vehicle emissions differ at different speeds, and (b) because at lower speeds the influence of the emissions from surrounding vehicles is higher.

3.6.3 Inner-vehicle distance

The inner-vehicle distance can influence the exposure levels since the reduced distance between vehicles at lower speeds in urban traffic, or when congestion is present on highways, allows emissions to infiltrate into vehicles nearby, resulting in higher exposures. In the study of McNabola et al. (2009b) a significant drop in in-vehicle VOC and $PM_{2.5}$ concentrations occurred within the first 2 m of their emission from the preceding vehicles exhaust.

3.6.4 Ventilation

Ventilation rates, whether driven by fans, natural leakage or open windows describe how rapidly outdoor air is capable of entering passenger cabins (Knibbs et al., 2011). Evidence suggests that ventilation is a key determinant of in-cabin concentrations in cars and buses as a high ventilation rate allows outdoor pollutants to enter the cabin (Zuurbier et al., 2010). Another constraint is the filtration system of the vehicle that helps to prevent ingress of particles, so that the vehicle is insulated against much of air pollution present in the street (Briggs et al., 2008). Recently, Querol et al. (2012) concluded that ventilation system and the air conditioning inside the trains decisively improve air quality in the subway system.

3.6.5 Fuel type

We identified only two studies where the fuel type was examined as an influencing parameter in personal exposure (Jalava et al., 2012; Zuurbier et al., 2010). Zuurbier et al. (2010) found the highest median particle number concentration and PM_{10} exposures in diesel cars and buses while the median soot exposure was highest in gasoline-fuelled cars. In that same study was reported that in electric buses exposure levels were lower than in diesel buses although difference for PM_{10} was small. However the observed differences were not statistically significant. Additionally, it is difficult to separate the effects of fuel type from those due to differences in ventilation under a standard setting between vehicles of different manufacturer (e.g. Knibbs et al., 2009).

In the study of Jalava et al. (2012) the PM emissions from a heavy-duty EURO IV diesel engine powered by three different fuels, a conventional diesel fuel and two biodiesels (methyl ester and hydrotreated vegetable oil) were evaluated and the toxicological properties of the emitted PM were investigated. The vegetable oil performed very well in emission reduction and in lowering the overall toxicity of emitted PM, but mixtures with the conventional diesel fuel were no better in this respect than the plain diesel fuel.

3.6.6 Meteorological conditions

Personal exposure concentrations in non-motorised forms of transport (cycling) are influenced to a higher degree by wind speed, whereas personal exposure in motorised forms of transport is driven to a higher degree by traffic congestion (McNabola et al., 2008 and 2009a). Duci et al. (2003) and Alm et al. (1999) found that the meteorological parameters that affect the passengers' exposures to CO and fine particles are wind speed (decreasing exposure) and relative humidity (increasing exposure). The analysis of the $PM_{2.5}$ exposure in London showed that wind speed has a significant influence on personal exposure levels. The occurrence of higher wind speeds is strongly associated with a decrease in personal exposure levels (Adams et al., 2001b). VOCs concentrations inside buses were highly associated to changes in meteorological conditions with higher concentrations being registered during lower temperatures and wind velocities (Parra et al., 2008). The type vehicle was investigated by conducting a small experiment with 2 different types of vehicles an electric car and a gasoline-fuelled car. The results showed that the particle number concentrations depend on the outdoor concentrations and by the air exchange rate of the vehicle. No difference was observed when the conventional vehicle was following the electric one since the majority of the surrounding vehicles are diesel fuelled.

3.6.7 Electric vs. gasoline vehicle exposure levels

Within the purposes of the present literature review an experiment was conducted to investigate the influence of other parameters like the vehicle type (electric or conventional) on the exposure levels. The vehicles under investigation were provided voluntarily for this study. A gasoline vehicle (Saab 93) and an electric car (Tazzari Zero 100% electric, model CLASSIC 2010 edition, ENISOLA S.L) were tested in terms of particle number and BC concentrations in a real-world urban circuit in Barcelona. Each vehicle was equipped with a DiscMini (Matter Aerosol) to monitor the particle number concentrations and a micro-aethalometer (Model AE51, Magee Scientific) to monitor BC levels. The instruments were placed in the car cabin approximately 1 hour before driving. From the 1-h measurements during stable conditions (the two cars were parked in parallel with the engines switched off and no passengers inside) the difference in the air exchange rate between the two vehicles was tested. As illustrated in Figure 1, both particle number and BC concentrations were higher inside the electric vehicle during stable conditions. Since no indoor emissions sources

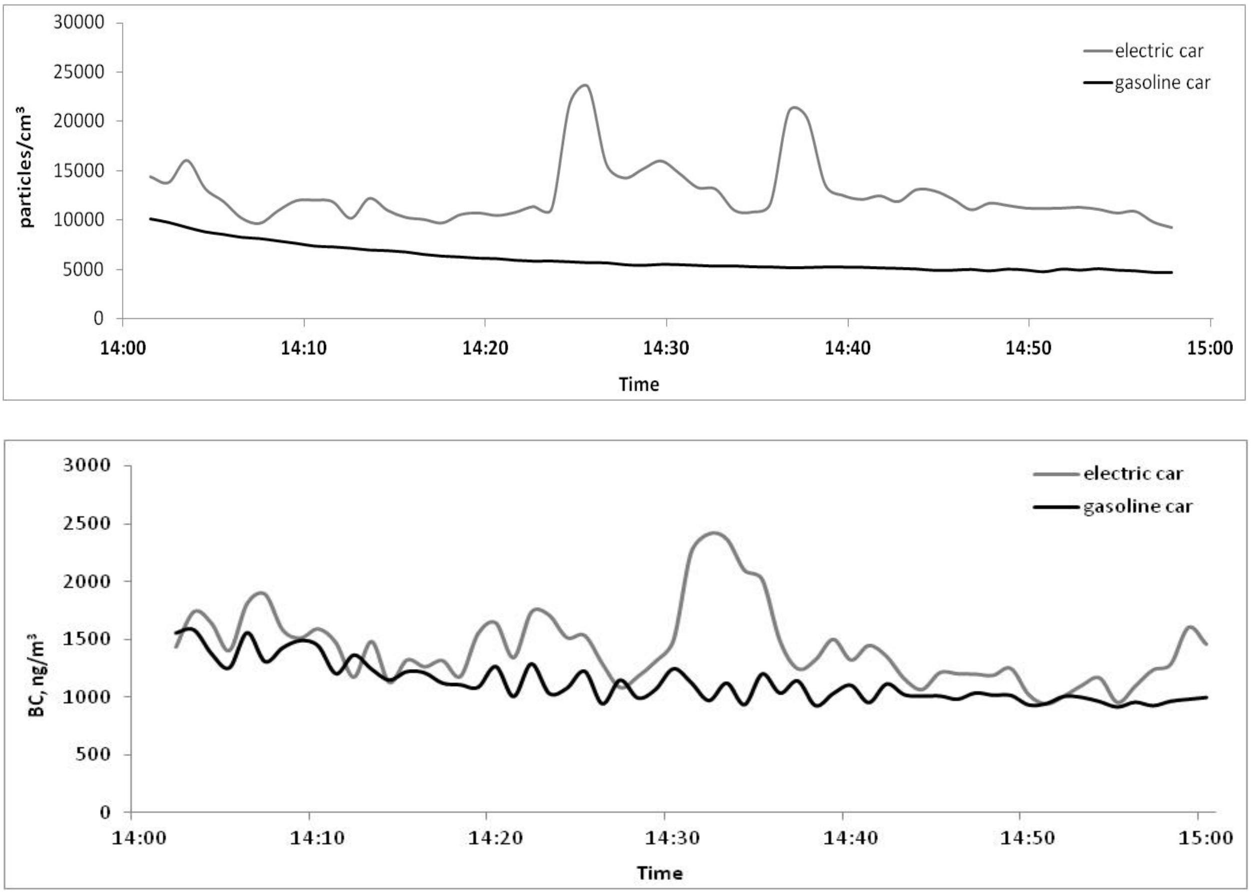
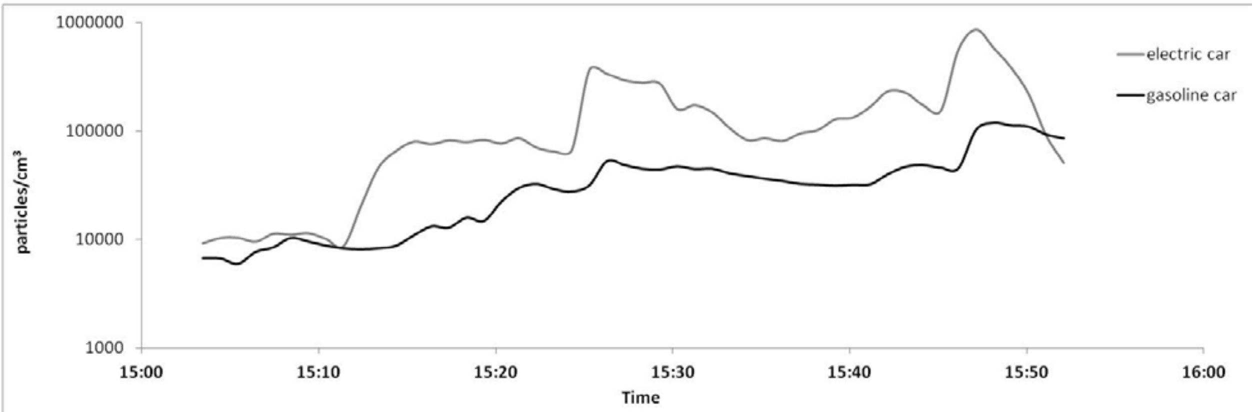


Figure 1. Particle number concentrations (top) and BC levels (bottom) for the two cars during stable conditions (engines off).

were present in the two cars and indoor concentrations were depending on the outdoor pollution levels this fact reveals the higher exchange rate of the electric car compared to the gasoline car. This result aimed to test the different infiltration rates in each car, which are only dependent on the vehicle structure and building materials.

Afterwards the volunteers conducted a 45 minute trip during heavy traffic conditions with the windows closed and two passengers in each car. Air re-circulation and air conditioning were switched off in both vehicles. In the first 30 minutes the two cars were moving in parallel. Although the concentrations levels were constantly higher inside the electric vehicle (due to the higher air exchange rate) the time variation was similar in the two cars revealing the strong influence of the outdoor (traffic) emissions. In the last 15 minutes of the trip the gasoline car drove directly behind the electric one in order to observe any reduction in the particle number and BC levels inside the gasoline car due to zero exhaust emissions of the electric car. Again no significant reduction was observed probably due to the higher influence of the emissions of the surrounding conventional vehicles, Figure 2.



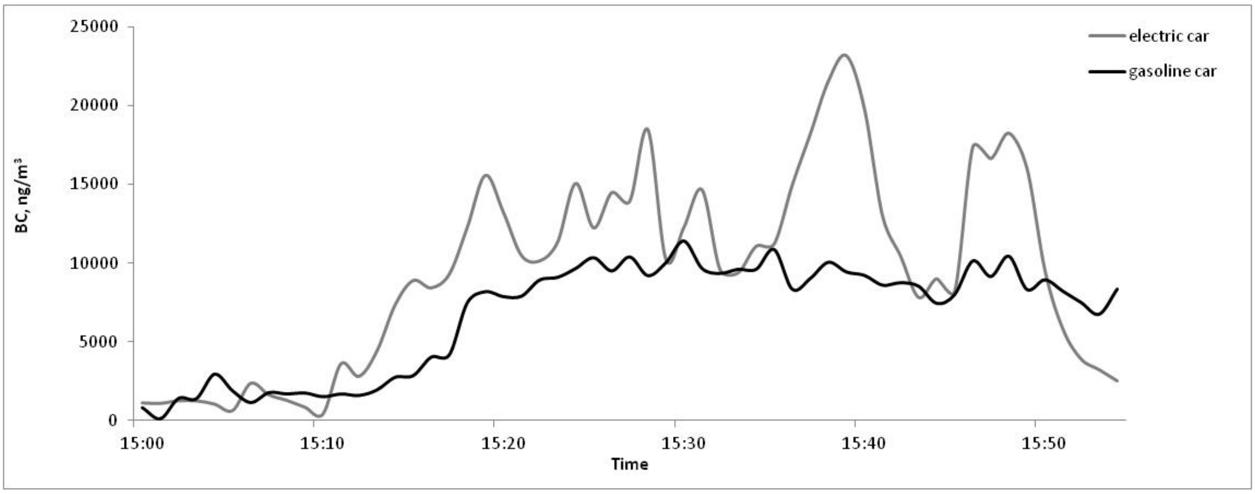


Figure 2. Particle number concentrations (top) and BC levels (bottom) for the two cars during driving in heavy traffic real-world conditions in an urban circuit.

Both results suggest that exposure during commuting by car depends on the emissions from the surrounding vehicles, rather than on the actual vehicle the subject is commuting in. Therefore, the composition of the vehicle fleet (clean or high polluting vehicles) plays a key role in exposure by car in urban environments, as do traffic intensity. It is interesting to observe that these results do not coincide with other studies found in the literature

(Zuurbier et al., 2010; Jalava et al., 2012), which suggest that pollutant concentrations inside the car depend highly on the type of fuel used. At this point we are unable to explain these different results.

4 Main conclusions and recommendations

Studies on personal exposure to air pollutants during **car commuting** are more numerous than those dealing with other types of transport, and typically conclude by emphasising that travelling by car involves exposure to relatively high PM exposure concentrations. Thus, compared to other transport methods, travelling by car has been shown to involve exposure both to higher PM (11%, according to Boogaard et al., 2009) and BC (Adams et al., 2002; De Nazelle et al., 2012) as compared with cycling. Car travel similarly involves inhalation of enhanced levels of CO (De Bruin et al., 2004; Vellopoulou et al., 1998; Dor et al., 1995), VOCs (McNabola et al., 2008) and particle number concentrations (Diapouli et al., 2008), especially during heavy traffic intensity. The pollutant exposure will however differ greatly depending on traffic intensity and speed and the type of ventilation inside the car. The need for reducing the numbers of diesel vehicles inside cities is evident and an obvious recommendation for the urban environment worldwide.

Personal exposure to PM for **bus commuters** depends on several variables, varying greatly from city to city, depending on the traffic intensity of each place and the type of buses in use. Thus, as in the case of travelling by car, levels of PM, BC and VOCs to which bus passengers are exposed will very much depend on the selected route, as highly busy streets will contain higher ambient levels of exhaust emissions from neighbouring vehicles (Dor et al., 1995; Barreford and Petreson 1996; Parra et al., 2008). Similarly, the contamination emitted by buses will be small in those cities where the bus fleet is modern and ecologically friendly, fitted with natural gas, hybrid, or diesel with filter trap systems (Zuurbier et al., 2010). As with the case for reducing diesel cars, the argument for improving urban bus fleets to reduce ambient air pollution is encouraging the implementation of modernising programmes in all cities.

Commuting by **underground rail** is a transport mode used daily by over one hundred million people worldwide, with more than 60 cities in Europe alone utilising rail subways to facilitate commuter movement. Although less information is available, studies on commuting in the subway show higher PM levels compared to other modes of transport (Nieuwenhuijsen et al., 2007). However, some subway systems present lower PM exposure levels than those shown by other transport modes (Chan et al., 2002a and 2002b). Of special concern is the fact that, when compared to outdoor air, subway air might be anomalously rich in metals, especially iron but also trace metals such as manganese, chromium, copper, nickel, zinc as well as the toxic metalloids antimony and arsenic (Querol et al., 2012). Key factors influencing subway PM concentrations include station depth, station design, type of ventilation (non-forced/forced), air-conditioning inside trains, types of brakes (electric/conventional brake pads) and wheels (rubber v. steel) used on the trains, train frequency, driving behaviour and more recently the presence or absence of platform screen door systems (Querol et al., 2012; Nieuwenhuijsen et al., 2007). These variables can lead to an increase of up to 8 times PM exposure when compare to other transport modes (Adams et al., 2001), especially on platforms rather than inside trains.

In general, exposure studies published so far have revealed that **cyclist commuters** are exposed to lower particulate matter concentrations in comparison to those inside vehicles.

However, when accounting for inhaled doses, this relationship is reversed, with cyclists having higher $PM_{2.5}$ lung deposition due to their greater energy expenditure and consequent inhalation rate (McNabola et al., 2008). The fact that proximity to the pollutant sources has a significant impact on exposure levels experienced by cyclists and pedestrians is an important aspect that should be included in developing traffic circulation plans, for example, by creating separate bicycle lanes parallel to the arterial roads feeding large volumes of traffic in and out of the city or accommodate bike lanes in quitter parallel streets (Jacobs et al., 2010; Yang et al., 2010; Puchewr and Buehler 2008). Even without employment of such avoidance strategies, however, the personal benefits of cycling are still significant. In this context, Rojas-Rueda et al. (2011) evaluated the health benefit of the bike sharing "*Bicing*" program in Barcelona, including the effect of pollution exposure for the cyclists (although not the public benefit due to reduced vehicle emissions). Public bicycle sharing initiatives such as *Bicing* are increasingly common and have wider benefits than simply the reduction of pollutant (Rojas-Rueda et al. 2011; Woodcock et al., 2009; Hartog et al., 2010; Rabl and deNazelle 2012). Foremost amongst these advantages, despite large uncertainties in the figures, is the health benefit gained by the individual due to the physical activity involved. It is this personal benefit, rather than the actual decrease in pollutant emissions induced by the switch away from car use, which perhaps provides the strongest case for encouraging urban bicycle transport.

All cities across the world face the considerable challenge of adopting transport policies that reduce, or at least do not enhance, ambient levels of air pollutants. Widespread dependence on private car transport has produced a significant daily health offence to the urban commuter, with consequent health risks that are especially threatening to those already compromised by illness or old age. However, enough has already been published to demonstrate that a forward-looking, integrated transport policy, involving the phased renovation of existing public vehicles and the withdrawal of the more polluting (especially diesel) private vehicles, combined with incentives to use public transport and the encouragement of commuter physical exercise, will produce immediate results towards solving what is, a difficult but not intractable problem.

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