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Review paper

A review of commuter exposure to ultrafine particles and its health effects

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ABSTRACT

Ultrafine particles (UFPs, <100 nm) are produced in large quantities by vehicular combustion and are implicated in causing several adverse human health effects. Recent work has suggested that a large proportion of daily UFP exposure may occur during commuting. However, the determinants, variability and transport mode-dependence of such exposure are not well-understood. The aim of this review was to address these knowledge gaps by distilling the results of 'in-transit' UFP exposure studies performed to-date, including studies of health effects.

We identified 47 exposure studies performed across 6 transport modes: automobile, bicycle, bus, ferry, rail and walking. These encompassed approximately 3000 individual trips where UFP concentrations were measured. After weighting mean UFP concentrations by the number of trips in which they were collected, we found overall mean UFP concentrations of 3.4, 4.2, 4.5, 4.7, 4.9 and 5.7×10^4 particles cm^{-3} for the bicycle, bus, automobile, rail, walking and ferry modes, respectively. The mean concentration inside automobiles travelling through tunnels was 3.0×10^5 particles cm^{-3}.

While the mean concentrations were indicative of general trends, we found that the determinants of exposure (meteorology, traffic parameters, route, fuel type, exhaust treatment technologies, cabin ventilation, filtration, deposition, UFP penetration) exhibited marked variability and mode-dependence, such that it is not necessarily appropriate to rank modes in order of exposure without detailed consideration of these factors. Ten in-transit health effects studies have been conducted and their results indicate that UFP exposure during commuting can elicit acute effects in both healthy and health-compromised individuals. We suggest that future work should focus on further defining the contribution of in-transit UFP exposure to

total UFP exposure, exploring its specific health effects and investigating exposures in the developing world.

Keywords: air pollution; transport modes; acute health effects; travel; public transport

1. INTRODUCTION

The study of commuter exposure to traffic-related air pollutants is not a particularly new field of research. Among the first researchers to recognise its significance was Professor Arie Haagen-Smit, who is best-known for his pioneering and enduring work related to characterising photochemical smog and ozone. In 1966, he performed a series of carbon monoxide measurements on heavily trafficked Los Angeles roads (Flachsbart, 2007; Haagen-Smit, 1966). Given population growth and increased motor vehicle use since that time, coupled with the high degree of proximity to vehicle emissions when commuting, the issue of 'in-transit' exposure to air pollutants is of equal if not greater relevance 45 years later.

Previous reviews of in-transit pollutant exposure, of which there are few, have focussed on CO inside vehicles (El-Fadel and Abi-Esber, 2009), particle mass concentrations and composition in metro (subway) systems (Nieuwenhuijsen et al., 2007) and various pollutants in multiple transport modes (Weisel, 2001). Only the work of Kaur et al. (2007) included a review of ultrafine (<100 nm) particle (UFP) concentrations in several transport modes.

At present, although gaseous pollutants are still the focus of numerous in-transit exposure studies, UFPs are beginning to attract significant attention. They are produced in large quantities by fuel combustion, and have been identified as a causal component of various negative health effects in humans (Knol et al., 2009; Hoek et al., 2010). UFPs typically constitute ~90% or more of particle number count (PNC) in areas influenced by vehicle emissions (Morawska et al., 2008), and we use UFP to describe PNC throughout this article.

The primary aim of this review is to distil the results of work performed to-date in order to improve understanding of the measurement, characteristics and determinants of in-transit exposure to UFPs, prior to a discussion of gaps in knowledge and suggestions for future research. Here, we extend the work of Kaur et al. (2007) by confining our focus to UFPs and incorporating the substantial body of relevant work that has appeared in the 4 years since its publication, which now constitutes the large majority of available literature. Like Kaur et al. (2007), we restrict our investigation to UFP exposure concentrations, rather than average or integrated exposure for a given time period. We note that dose assessment, which is a complementary yet distinct concept to that of exposure (Ott, 1985), is not the main focus of this review.

This review begins with an overview of the nature of commuter travel prior to a description of the general characteristics of UFPs. This is followed by a detailed analysis of in-transit UFP exposure studies, a discussion of determinant factors, health effects, and suggestions for future research.

1.1 Commuting in modern society

The nature of modern society in many countries both affords and expects a high degree of personal mobility. Time-activity patterns define how people apportion their time across a range of environments, and are a keystone of effective exposure analyses. Time-activity pattern studies of varying magnitude performed in different regions have reported that time spent in-transit typically constitutes between about 5 and 10% of the day (Klepeis et al., 2001), depending on location. The transport microenvironment(s) within which this time is spent varies more substantially between regions than the occupancy time, and has a greater dependence on local factors, such as the availability and desirability of various transport options.

In general, there are scant 24 hour time-activity pattern data for developing countries. Saksena et al. (2007) reported that time spent travelling among 4311 Delhi residents ranged from 0.8 to 10% of the day, and varied markedly depending on age, sex and occupation, as did the mode of transport used. It is likely that the time-activity patterns of people in rural areas differ significantly from those of their urban counterparts.

1.2 Children's and adult's travel choices

Children are particularly susceptible to negative health effects caused by exposure to air pollutants (Gauderman et al., 2004; Brugge et al., 2007; Ashmore and Dimitroulopoulou, 2009), and many millions are required to commute between home and school each weekday. The choice of which transport mode school children utilise is normally at the discretion of others. Whilst children and young people have been reported to possess informed and responsible opinions regarding transport choices and a clear ideal towards cycling and walking, their parent's choices are guided primarily by safety concerns, and place substantial reliance on private automobiles (Lorenc et al., 2008).

Unlike children, adults generally make their own travel choices. A recent survey of 745 employed adults in Queensland, Australia, found that while about half of respondents felt that exposure to air pollutants in-transit negatively affected their overall health and increased their risk of cardiovascular disease, only 13% indicated that exposure to pollutants was a barrier to their adoption of walking or cycling to work, and that other factors were more responsible for their high level (82%) of dependence on private transport (Badland and Duncan, 2009). Furthermore, Badland and Duncan (2009) found that adults who were better educated and lived in urban areas were most cognisant of the negative health effects of air pollutant exposure during transit. Marshall et al. (2009) reported that the optimum balance between high walkability and low pollution was identified sporadically and typically in higher income neighbourhoods in urban Vancouver (Marshall et al., 2009). Evidently, there may be a significant socio-economic component involved in air pollution exposure during transit, particularly for active transport modes, and this may reflect wider socio-economic and environmental inequalities reported for several traffic pollutants (Marshall, 2008; Tonne et al., 2008; Su et al., 2009). It should be noted that both children and adults in developing countries are unlikely to be afforded the luxury of a travel choice, *per se*, and a relatively high degree of dependency on walking and public transport may result from this (Saksena et al., 2007).

2. CHARACTERISTICS OF ULTRAFINE PARTICLES

2.1 General

UFP concentrations reflect the contribution of anthropogenic processes to a pre-existing background concentration (Morawska et al., 2008). Background concentrations are ascribed to natural processes, such that in most environments free from the immediate influence of anthropogenic activities, UFPs are present and their concentrations readily measured. Despite the numerous natural sources of UFPs, vehicular fossil fuel combustion has repeatedly been shown to be their dominant source in urban areas, with heavy-duty diesel powered vehicles making a disproportionately large contribution to UFP concentrations (Morawska et al., 2008).

An important distinction is between primary and secondary UFPs. The primary variety are emitted from their source as particles, whilst secondary particles are formed following homogenous nucleation of gases (Koutrakis and Sioutas, 1996; Jacobson, 2002). This occurs when a gas, or gases, nucleate in the absence of a pre-existing surface (Jacobson, 2002).

UFPs from vehicles can be emitted as primary particles or generated as a secondary aerosol, often following homogenous nucleation of SO_2, NH_3 and NO_x into SO_4^{2-}, NH_4^+ and NO_3^- (Koutrakis and Sioutas, 1996; Morawska et al., 2008). The ratio of primary to secondary particles varies substantially according to fuel type and operating and environmental conditions, but nucleation mode particles can comprise approximately 90% or more of UFPs in diesel exhaust (Kittelson, 1998). However, more recent research indicates that the number of nucleation mode particles in diesel exhaust can be reduced to 40-50% when ultra-low sulphur diesel fuel is used (Ristovski et al., 2006), which is more representative of modern vehicle fleets in many countries.

2.2 Typical concentrations

Morawska et al. (2008) performed a meta-analysis of 71 UFP studies performed across a diverse range of environments. They found mean concentrations of 2.6, 4.8, 7.3, 10.8, 42.1, 48.2, 71.5 and 167.7 × 10^3 particles cm^{-3} for clean background, rural, urban background, urban, street canyon, roadside, on-road and tunnel environments, respectively. This indicates that greater proximity to vehicles is associated with increased UFP concentrations, and underscores their importance as a UFP source.

2.3 Health significance

Once inhaled, UFPs can reach with the alveolar region of the human lung with greater efficiency than larger particles due to their smaller size, and can deposit in alveoli with greater efficiency as a consequence of their rapid diffusion (Daigle et al., 2003; Phalen et al., 2006; Frampton, 2007). Due to their content of reactive oxygen species (ROS) and large combined surface area, UFPs from vehicle emissions have the potential to damage pulmonary cells (Delfino et al., 2005). Transition metal components in UFPs are believed to play a role in producing ROS, along with pro-oxidative organic hydrocarbons (Li et al., 2003). Additionally, target cells, such as airway epithelial cells and macrophages, produce ROS during biologically catalysed redox reactions occurring in the mitochondria in response to UFP uptake (Li et al., 2003; Nel, 2005). UFPs can evade alveolar macrophage clearance from the lung and enter lung cells, the interstitium and possibly the vascular bed (Geiser et al., 2005; Frampton, 2007), and can travel from the lung via blood and lymphatic circulation to other organs (Elder et al., 2006; Samet et al, 2009). UFPs are more proatherogenic than larger particles due to their greater bioavailability of reactive compounds, content of redox-active compounds, high number concentration and increased lung retention (Aruajo et al., 2009).

Epidemiologic investigations of UFPs have been constrained by the scarcity of UFP monitoring sites and the substantial spatial heterogeneity of concentrations (Brook et al., 2010). Studies performed to-date in Erfurt, Germany, have indicated that UFP effects on daily mortality may be of comparable magnitude to, yet independent of, those of fine particles (i.e. $PM_{2.5}$), albeit with greater time lag between UFP concentrations and their effects (Wichmann and Peters, 2000). More recent results from the same long-term study have shown statistically significant associations between UFP concentrations and both total and cardio-respiratory daily mortality with a four day lag period (Stölzel et al., 2007).

Interestingly, this study found no association between $PM_{2.5}$ mass concentration and mortality. Mortality from stroke amongst aged residents of Helsinki during summer was associated with both $PM_{2.5}$ and UFP concentrations on the previous day, and effects were mostly independent (Kettunen et al., 2007).

The effects of UFP concentration on mortality and morbidity due to various causes are less well understood than those of larger particles. A recent elicitation of European experts found that short-term UFP exposure was rated to variously possess a medium to very high likelihood of causality for all-cause mortality, and a low to high likelihood for cardiovascular and respiratory hospital admissions (Knol et al., 2009). Long-term UFP exposure was generally rated to possess a slightly lower likelihood of causality for all-cause mortality, owing mainly to the lack of long-term studies (Knol et al., 2009). The same group of experts estimated that a permanent decrease in annual average UFP concentration of 1000 particles cm^{-3} across Europe would lead to median decreases of 0.3%, 0.2% and 0.16% in all-cause mortality, and cardiovascular and respiratory hospital admissions, respectively (Hoek et al., 2010). The relatively small number of epidemiological studies (14) and absence of long-term studies, however, resulted in most experts indicating a substantial degree of uncertainty in their estimates (Hoek et al., 2010).

3. STUDIES OF UFP CONCENTRATION IN TRANSPORT MODES

3.1 Methods

We searched combinations of the terms “ultrafine particle”, “transport mode”, “commuter”, “exposure” “public transport”, “microenvironment”, “vehicle”, “car”, “automobile”, “bus”, “cyclist”, “bicycle”, “train”, “metro”, “subway” on PubMed, ISI Web of Knowledge and Google Scholar until October, 2010. The reference lists of studies identified by this method were reviewed for links to additional literature. Furthermore, the authors' own literature collections were utilized.

We restricted our investigation to studies that presented numeric values of UFP concentrations, and identified 47 that fulfilled this requirement. Tables S1-S7 in the Supplementary Information file contain detailed information on the various studies. These spanned 6 distinct transport modes: automobile, bus, cycling, ferry, rail and walking. Some studies dealt with multiple transport modes, whilst others focussed on a single mode. Of the studies we identified, only 7/47 (15%) had previously been reviewed by Kaur et al. (2007), which highlights the rapid progression of research related to in-transit UFP exposure since publication of their work.

The mean concentrations extracted from the studies identified were weighted by the corresponding number of trips taken, and overall trip-weighted mean UFP exposure concentrations were calculated for each transport mode (see tables S1-S7 in the Supplementary Information file). The overwhelming majority of studies (93%) reported the number of trips associated with a given mean; the means reported by those that did not report trip number were weighted by a conservative factor of 1 trip. Most studies reported arithmetic mean UFP concentration, while several reported geometric mean and one gave median values. Where possible, data were disaggregated to permit analyses of the effect of variables such as fuel type, presence of exhaust-treatment devices and route.

Given the range of conditions under which they were collected, we did not assess the statistical significance of differences in measured mean UFP concentrations between modes, and instead sought to identify general trends in the data. This is discussed further in sections 4 and 5.

3.2 Results

Across all modes, we identified approximately 3000 individual trips where UFP measurements were performed. There was an uneven distribution of measurement trips; very few have been performed in ferry (13) and rail (49) modes, while a substantial number have been undertaken in bus (505), walking (524), cycling (599) and automobile (1310) modes. The automobile mode was split into non-tunnel (977) and tunnel (333) trips, as previous results indicate that tunnels are a discrete UFP exposure microenvironment distinct from open air roadways (Kaminsky et al., 2009; Knibbs et al., 2010).

Figure 1 shows the trip-weighted mean UFP concentrations for each mode, and the number of trips on which they were based. Error bars indicate the trip-weighted standard deviation (Bland and Kerry, 1998). The range of mean UFP concentrations spanned one order of magnitude, with the lowest measured whilst cycling and the highest in automobiles during tunnel travel; 3.4×10^4 (s.d. = 1.8×10^4) and 3.0×10^5 (s.d. = 2.6×10^5) particles cm^{-3}, respectively. Means and standard deviations calculated for the automobile (non-tunnel), bus, ferry, rail and walk modes were 4.5 (3.3), 4.2 (3.1), 5.7 (0.5), 4.7 (4.1) and 4.9 (3.2) $\times 10^4$ particles cm^{-3}, respectively.

FIGURE 1 TO BE INSERTED HERE

4. COMPARISON BETWEEN MODES

Considering the diversity of studies from which they were drawn, the trip-weighted concentrations measured in automobile (non-tunnel), cycle, bus, rail and walking modes exhibited notable coherence, with a maximum to minimum ratio (walk:cycle) of 1.5. A limited number of studies that measured concentrations in different modes simultaneously or near-simultaneously have been reported, and Briggs et al. (2008) found a walk:automobile ratio of 1.4, which is higher than the value of 1.1 presented here. Boogaard et al. (2009) found an automobile:cycle ratio of 1.05, whilst we found a value of 1.3, which was also higher than the value of approximately 1.0 reported by Int-Panis et al. (2010).

While the above studies highlighted the relative concentrations encountered in each mode in the absence of bias due to fluctuating UFP concentrations, observed inter-mode contrasts were specific to the conditions of the study (e.g. the ventilation settings in an automobile, or the proximity to traffic on a bike route) and are should therefore not be extrapolated beyond the conditions under which they were collected without appropriate caution.

In studies that measured UFP concentrations in multiple modes non-simultaneously, the mode in which highest concentrations were recorded vacillated between automobiles and buses, whilst those in other modes were markedly lower (Levy et al., 2002; Kaur et al., 2005b; Weichenthal et al., 2008; Cattaneo et al., 2009; Kaur and Nieuwenhuijsen, 2009; Pattinson, 2009; Shrestha, 2009; Knibbs and de Dear, 2010). It is therefore noteworthy that our analysis found that UFP concentrations in buses and automobiles (non-tunnel) were relatively low. We pooled a large number of reported UFP measurements performed under a wide range of conditions, and while the ability to differentiate the observed differences is limited by the level of detail given by the various studies, such an approach is indicative of mean values and general trends. However, the mode in which highest exposures are experienced depends strongly on the determinant factors discussed in the following two sections, and generalisation of results may be of limited value (Int-Panis et al., 2010); that is, within mode variability is likely to be substantial.

5. DETERMINANTS OF UFP CONCENTRATION IN-TRANSIT

Despite the convenience it may provide, it is not necessarily appropriate to rank transport modes in order of UFP exposure without certain caveats. For example, figure 1 shows the trip-weighted mean UFP concentration in an automobile is higher than the equivalent for cycling. However, an occupant of a relatively air-tight automobile in which air is recirculated and filtered will likely experience markedly lower exposure concentrations than a cyclist on a high traffic route. Disentangling the relative roles of determinant factors, their interactions and variability in each mode is a key element required to advance understanding of in-transit UFP exposure. The data reviewed here suggest that while the relationship between UFP concentration and its determinants is often mode-dependent, exposure in all mode types is the result of interplay between multiple factors. These can be viewed as comprising two stages: the first determines the outdoor or on-road UFP concentration, and the second determines what proportion of this is able to come into contact with a commuter. These factors are addressed, in turn, in the following sections.

5.1 Meteorological variables

Temperature has been variously reported to be positively and negatively correlated with UFP concentrations, although in vehicle-dominated areas the correlation is more likely to be negative due to condensation of volatile compounds in emissions (Morawska et al., 2008). In-transit studies that assessed this relationship uniformly found a negative correlation between temperature and UFP concentration (Krausse and Mardaljevic, 2005; Vinzents et al., 2005; Thai et al., 2008; Weichenthal et al., 2008; Kaur and Nieuwenhuijsen, 2009; Pattinson, 2009; Laumbach et al., 2010). Among studies that reported correlation coefficients, those measured for cycling studies (Vinzents et al., 2005; Thai, 2008) were quite high (-0.62 and -0.76, respectively). Multi-mode and automobile studies reported correlations of -0.77 and -0.37, respectively (Kaur and Nieuwenhuijsen, 2009; Laumbach et al., 2010).

Wind speed, which affects dilution and transport of vehicle emissions, was also found to be negatively correlated with UFP concentration in-transit (Krausse and Mardaljevic, 2005; Vinzents et al., 2005; Briggs et al., 2008; Thai et al., 2008; Weichenthal et al., 2008; Kaur and Nieuwenhuijsen, 2009; Pattinson, 2009; Srestha, 2009; Knibbs and de Dear, 2010), which is in agreement with results reported for various outdoor locations (Morawska et al., 2008). However, the results were not always statistically significant, indicating that temperature may be more consistently and strongly correlated with UFP concentrations. Correlations observed for active transport modes were -0.20 (walk), -0.52 (cycle) and -0.81 (cycle) (Briggs et al., 2008; Vinzents et al., 2005; Thai et al., 2008). Multi-mode and automobile studies reported correlations of -0.14 to -0.49 (Briggs et al., 2008; Kaur and Nieuwenhuijsen, 2009; Knibbs and de Dear, 2010).

For both temperature and wind speed, stronger correlations were generally observed for cycling compared to non-active modes, perhaps reflecting reduced influence of other factors on exposure concentrations encountered when cycling (and walking). The strength of the association between UFP concentration and both temperature and wind speed appears to be mode and location-dependent, and its variability is not well characterised.

While temperature and wind speed are the most frequently reported, other meteorological parameters may affect UFP concentration. The depth of the mixed layer within the atmosphere was found to be negatively correlated with in-transit UFP concentration (Weichenthal et al., 2008), which reflects the tendency of a shallow mixed layer to concentrate pollutants.

5.2 Traffic volume and composition

Very few studies have reported the relationship between traffic volume and in-transit UFP concentrations. Fewer still have examined the effect of traffic composition (i.e. gasoline vehicles, diesel vehicles). Briggs et al. (2008) observed statistically significant correlations between car and truck density and UFP concentrations encountered while walking (r = 0.41 to 0.48) or in an automobile (r = 0.43 - 0.47) in London. In their London-based study, Kaur and Nieuwenhuijsen (2009) similarly found a significant correlation (ρ = 0.27) between total traffic count and UFP concentrations in automobile, bus, cycle, taxi and walking modes. Krausse and Mardaljevic (2005) reported road link end description (e.g. signal, left turn, right turn etc.) was a significant determinant of total UFP exposure of car occupants. On-road studies have shown strong associations (R^2 ~0.85) between heavy diesel traffic volume and UFP concentrations (Fruin et al., 2008; Knibbs et al., 2009b). Other studies have reported more qualitative assessments of traffic effects; for example, that mean in-transit UFP concentrations increased on high traffic routes and vice-versa (Zhu et al., 2007; Thai et al., 2008; Strak et al., 2010; Zuurbier et al., 2010).

Vehicle emissions are the dominant source of UFPs in urban areas, and heavy diesel vehicles make a contribution that is disproportionate to their volume (Morawska et al., 2008). Coupled with the limited but consistent findings of in-transit studies, this suggests that traffic parameters (volume, density) and composition (gasoline vehicles, heavy diesel vehicles) are an important determinant of in-transit UFP exposure. It should be considered, however, that effects are likely to depend on mode, and that short-term traffic patterns not represented in hourly or daily average data, such as the impact of passing traffic, may be important (Fruin et al., 2008; Boogaard et al., 2009).

5.3 Route choice: active transport modes

There are generally fewer mode-specific variables that may affect pedestrians and cyclists compared to other transport modes; that is, traffic and meteorological conditions may be of greater importance as determinants. Most cycling studies were performed on or proximate to major urban roads, however, some studies compared measurements on high and low traffic routes, with the latter typically comprised of a dedicated cycle path. Separating the data into these two categories revealed that 18% of trips were undertaken on low traffic routes, and mean UFP concentrations were 2.6×10^4 particles cm^{-3}. The mean for high traffic routes was 3.5×10^4 particles cm^{-3}, suggesting that route selection, within the context of the few studies to address it, can affect cyclist UFP exposure (Pattinson, 2009; Strak et al., 2010; Zuurbier et al., 2010).

Route choice, as a proxy for traffic volume, is likely to be an important determinant of exposure (McCreanor et al., 2007; Hertel et al., 2008), and personal factors (e.g. walking or cycling patterns) may also exert an effect (Kaur et al., 2007). Microscale variations in UFP concentration proximate to roadways may result in higher exposures on the road side of the sidewalk/footpath compared to the building side (Kaur et al., 2005a). Also, the effect of roadway factors on pollutant dispersion (i.e. whether open to the environment or prone to trap pollutants due to geometry of urban canyons) has been shown to be a statistically significant determinant of UFP exposure concentrations encountered when walking (Briggs et al., 2008). Further work focussed on evaluating the effects of these local and microscale route phenomena on UFP exposure is required.

5.4 Cabin ventilation

Ventilation rates, whether driven by fans, natural leakage or open windows (Ott et al., 2008; Knibbs et al., 2009a), describe how rapidly outdoor air is capable of entering passenger cabins. Evidence suggests that ventilation is a key determinant of in-cabin UFP concentrations in automobiles, buses (Hammond et al., 2007; Rim et al., 2008; Knibbs and de Dear, 2010; Zhang and Zhu, 2010), ferries (Hill et al., 2007; Knibbs and de Dear, 2010) and rail modes (Hill et al., 2007; Cheng et al., 2009; Knibbs and de Dear, 2010). Quantitative studies support these observations, but are scarce and limited to automobiles (Xu and Zhu, 2009; Knibbs et al., 2010).

Knibbs et al. (2009a) found that air exchange increased linearly with vehicle speed in a group of six test automobiles operating under four distinct ventilation settings, which was in agreement with results obtained by Ott et al. (2008) based on tests performed in four vehicles. Knibbs et al. (2009b; 2010) found that the primary determinant of on-road UFP concentration in a tunnel bore was hourly heavy diesel vehicle volume (R^2 = 0.87) , and that cabin ventilation rates explained 81% of the variation in the proportion of on-road UFPs reaching the occupants of 5 automobiles. The proportion reaching the cabin varied from 0.08 (recirculation) to ~1.0 (non-recirculation) depending on vehicle and ventilation setting. Thus, ventilation rates controlled the extent to which in-cabin exposure concentrations reflected on-road levels in the tunnel bore, which were largely determined by heavy diesel vehicle volume. Xu and Zhu (2009) reported that cabin ventilation and leakage were predominant factors in their model-based analyses of variables affecting in-cabin/on-road (I/O) ratios, and explained up to ~60% of on-road UFP ingress. I/O ratios when windows are open can reach 1 due to higher air exchange, and such conditions may also occur when windows are closed but ventilation fan settings are high (Ott et al., 2008; Knibbs et al., 2009a).

Some investigators have successfully performed in-cabin UFP size distribution measurements during transit in automobiles (Zhu et al., 2007) and buses (Zhang and Zhu, 2010). These studies have shown that while in-cabin particle size distributions follow the general shape of those on-roads, the ability of on-road particles to reach the cabin is dependent on particle size and ventilation settings (Zhu et al., 2007). Particle penetration is discussed in section 5.6.

5.5 Filtration

Where a vehicle is fitted with a cabin air filter, its efficiency is a key determinant of what proportion of on-road UFPs reach the cabin, and efficiency varies substantially amongst the filters available. Standard automobile cabin filters afford single-pass UFP reductions of between approximately 30 and 60% (Pui et al., 2008; Qi et al., 2008), while this can increased by employing more advanced filtration technologies (Burtscher et al., 2008). It should be noted that filtration efficiency is affected by the ventilation rate; as filter face velocity increases with mechanical or natural ventilation rates, filtration efficiency decreases due to the reduced time available for particle diffusion inside the filter (Pui et al., 2008; Qi et al., 2008). When air is recirculated in an automobile, Qi et al. (2008) found that UFP concentrations decayed most rapidly in a vehicle capable of filtering recirculated air (single pass efficiency = 46%) than in a vehicle lacking this feature, where UFP removal efficiency without a filter was 27% per recirculation of cabin air. In the former and latter cases, on-road UFP concentrations were reduced to those typical of an office building (4000 particles cm^{-3})

in 3 minutes and 9-10 minutes, respectively, indicating the value of recirculation as a simple exposure minimisation mechanism. However, some older, less-airtight vehicles are characterised by outdoor air exchange rates up to 47 hr^{-1} when air is recirculated (Knibbs et al., 2009a), and the benefit of recirculation in such cases can be substantially diminished (Knibbs et al., 2010).

5.6 UFP penetration and deposition

The penetration of UFPs through automobile envelopes is dependent on their size, the number and geometry of cracks, and the pressure difference across these and other ingress pathways (Xu et al., 2010). A recent study reported that penetration efficiency close to 100% was observed for diesel exhaust particles between 100 and 287 nm, and declined to ~70% for 10 nm particles due to diffusion; although penetration of 10 nm particles increased to ~90% when pressure differentials reached 200 Pa (Xu et al., 2010). No difference was observed in penetration efficiency amongst different materials.

Given the high surface to volume ratios of many automobile cabins, deposition can be an important UFP removal mechanism, especially under low ventilation conditions (Gong et al., 2009). Gong et al. (2009) found in-cabin deposition rates in automobiles exceed those of indoor environments by a factor of 3 to 20.

Studies describing UFP filtration, penetration and deposition in bus and rail modes are scarce and the limited data to-date is strongly skewed towards automobiles. Future studies addressing this knowledge gap will be of considerable value.

5.7 Fuel type and presence of an emission control device

Automobile: The effect of fuel type on UFP concentration in automobiles was assessed by Zuurbier et al. (2010), who found no significant difference in mean levels in diesel and gasoline-powered vehicles (diesel:gasoline concentration ratio = 0.96) based on 14 simultaneous trips under a standard ventilation setting. Their study focussed only on newer vehicles (< 6 months) and its relevance to the wider passenger vehicle fleet is unknown. 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., 2009a). Further studies involving test vehicle groups more representative of the heterogeneity present in wider vehicle fleets are required.

Bus: Due to their frequency of door opening and the 'stop-start' nature in which they travel, buses have a tendency to self-pollute (Behrentz et al., 2004; Hill et al., 2005; Rim et al., 2008; Liu et al., 2010; Zhang and Zhu, 2010; Zuurbier et al., 2010). Accordingly, the variables most frequently reported by UFP exposure studies were fuel type and the presence of an exhaust or crankcase emission control device. We therefore disaggregated bus trips into 8 categories: diesel, biodiesel, compressed natural gas (CNG), electric, diesel with oxidation catalyst (DOC), diesel with diesel particulate filter (DPF), diesel with crankcase filtration system (CFS), and diesel with combined control (i.e. any combination of two or more of DOC, DPF, CFS and ultra low sulphur diesel). About 70% of trips were performed in diesel buses, with the remainder approximately evenly distributed across the other categories. Five percent of bus trips (26/505) were excluded due to lack of detailed data on fuel type or control device.

Figure 2 shows the trip-weighted mean UFP concentrations for each category. The lowest mean (1.7×10^4 particles cm^{-3}; SD = 0.8×10^4) was recorded in CNG-powered buses, and the highest (4.9×10^4 particles cm^{-3}; SD = 2.6×10^4) was measured in diesel buses fitted with a CFS, although the latter result was based on a very limited number of trips (13). A similar mean was recorded in diesel buses with no control device (4.8×10^4 particles cm^{-3}; SD = 3.2×10^4). Means and standard deviations calculated for the biodiesel, combined control, DPF, DOC and electric categories were 1.7 (-), 2.0 (1.8), 2.4 (0.9), 2.8 (2.0) and 2.9 (0.8) $\times 10^4$ particles cm^{-3}, respectively. With the exception of the electric bus category, lowest concentrations were measured in buses powered by alternative fuels. Concentrations inside diesel-powered buses were generally lower when fitted with an emission control device.

FIGURE 2 TO BE INSERTED HERE

Differentiating the effects of self-pollution from those of other factors on in-bus UFP concentrations is challenging. Previous work has shown that self-pollution can be the dominant source of vehicle emissions in the cabin when windows are closed, and constituted 0.01 to 0.3% of air in the cabins of 1975 through 2002 model school buses (Behrentz et al., 2004). Liu et al. (2010) found that self-pollution contributed an overall average of 1.8×10^4 particles cm^{-3} in two school buses (2000 and 2003 model); the average contribution when windows were closed (1.0×10^4 particles cm^{-3}) was less than that when they were open (2.6×10^4 particles cm^{-3}). However, this trend was not in keeping with their results for other measured pollutants, and was attributed to UFP fluctuations due to unidentified non-vehicle sources on the low-traffic routes they studied. Generally, if on-road concentrations are low relative to those in-cabin, open windows will dilute self-pollution (Liu et al., 2010). The reverse can exacerbate its effects.

The relatively small number of trips taken in most categories we analysed and the lack of specific information regarding other possible determinants limits the conclusions that can be drawn, and precluded detailed statistical analyses. However, the results generally suggest that UFP concentrations are greatest in diesel-powered buses, and that reductions may be possible through use of alternative fuels or emission control devices, with best results achieved for diesel buses when two or more of the latter are combined.

Rail: In most rail studies we identified, trips were undertaken in vehicles driven by electricity. About 29% of trips were taken in diesel-powered trains, and the weighted mean UFP concentration during these was 9.0×10^4 particles cm^{-3}. The mean during travel in electric-powered vehicles was 3.0×10^4 particles cm^{-3}. Based on the limited data available, the power source of the rail vehicle therefore appears to affect UFP exposure concentrations. Moreover, in diesel trains, the position of the locomotive relative to the passenger compartments can markedly affect UFP concentrations; when a locomotive was located in front of passenger cabins, its emission plume can reach the cabin ventilation system intake, and vice-versa (Hill et al., 2007).

There was insufficient data to investigate the effect of underground and above ground travel on UFP concentrations. Whilst there are numerous mechanical processes that can generate and resuspend particulate matter in electric-powered subway/metro systems, these are more likely to elevate levels of particle mass rather than UFP number count (Nieuwenhuijsen et al., 2007). The limited number of studies reporting UFP measurements on underground platforms tend to support this (Aarnio et al., 2005; Seaton et al., 2005; Raut et al., 2009; Cheng et al., 2009; Nystrom et al., 2010).

6. CORRELATION WITH OTHER AIR POLLUTANTS

Several in-transit studies measured UFPs and other pollutants simultaneously. A summary of these is provided in Table S8 in the Supplementary Information file. The correlation between UFP and $PM_{2.5}$ concentrations is generally reported to be positive, weak and not statistically significant, although stronger associations have been observed; correlation coefficients range from -0.07 to 0.69 (Aarnio et al., 2005; Kaur et al., 2005a,b; Seaton et al., 2005; McCreanor et al., 2007; Zhu et al., 2008; Berghmans et al., 2009; Boogaard et al., 2009; Knibbs and de Dear, 2010; Laumbach et al., 2010). Although correlation in the rail mode is moderate and relatively consistent across studies, in general there is no clear relationship between the strength of correlation and transport mode. The results are likely to be somewhat location-dependent, in keeping with those for outdoor environments, and the generally poor correlation reflects differences in the sources of particle number and mass and temporal scales involved in their dynamics (Morawska et al., 2008).

Black carbon (BC) and elemental carbon (EC) are often well-correlated with UFP concentrations in urban air, given their shared provenance in vehicle emissions and the large extent to which BC and EC contribute to UFP chemical composition (Morawska et al., 2008). On-road and subway platform studies have shown very good correlation between UFPs and BC; 0.88 and 0.84, respectively (Aarnio et al., 2005; Westerdahl et al., 2005). Correlations were relatively weak in automobile and bus studies (mean = 0.1 to 0.2), although in-bus relationships were strongly dependent on window position, and mean correlation improved (mean = 0.62) when windows were kept open, which the authors ascribed to self-pollution under the closed window setting (Zhu et al., 2008; Zhang and Zhu. 2010). Very good correlations between UFPs and EC (0.70 and 0.84) have been reported in walking studies (Kaur et al., 2005a; McCreanor et al., 2007).

The correlation between UFP concentrations and those of NO_X vary extensively from -0.33 to 0.90, and no clear relationship with transport mode is apparent (Westerdahl et al., 2005; McCreanor et al., 2007; Zhu et al., 2008; Laumbach et al., 2010). The relationship with CO concentrations is similarly variable; -0.16 to 0.70 (Kaur et al., 2005a,b ; Westerdahl et al., 2005; McCreanor et al., 2007; Zhu et al., 2008; Laumbach et al., 2010). The specifics of the measurement location in terms of local emission sources are likely to explain the observed variation, and it is important to consider that in-transit measurements of particle and gaseous pollutants may exhibit poor temporal correlation due to the varying emission strength of proximate vehicles (Morawska et al., 2008; Zhu et al., 2008).

In summary, the relationship between in-transit UFP concentrations and those of other pollutants is generally inconsistent. Mode, location and environmental factors may all contribute to the observed variability, and the results gathered here from the limited pool of available studies require further validation in order to develop a more complete understanding of the associations. Currently, there is no data to support prediction of UFP concentrations from those of other pollutants, and such an approach is likely to be insufficient.

7. RELATIONSHIP WITH FIXED SITE MONITORS

Since the 1970s (Ott and Eliassen, 1973; Cortese and Spengler, 1976), numerous studies have investigated the ability of fixed site pollutant monitoring stations to estimate personal and commuter exposure. Generally, the ability of fixed site monitors to represent the substantial spatial and temporal variability of in-transit exposures has been sub-optimal, and carries with it numerous attendant limitations, the most important of which is underestimation of exposure (Kaur et al., 2007). UFPs are not a regulated pollutant, and are therefore not routinely monitored outside of research studies. Some investigators have assessed the association between fixed site UFP concentrations and those measured concurrently in-transit.

Aarnio et al. (2005) reported good correlation ($R^2 = 0.59$) between UFP concentrations in subway stations and those measured at an urban background site, while Seaton et al. (2005) found that the ratio of UFP concentrations measured on London Underground platforms to those above ground ranged from 0.38 to 0.68. These results are likely to reflect the absence of strong local UFP sources in subways (Aarnio et al., 2005). For above ground transport microenvironments, however, this is unlikely to be the case. Vinzents et al. (2005) reported a moderate correlation ($r = 0.49$) between UFP measurements performed at a fixed roadside location and those measured while cycling, but found that the only significant variables in a linear mixed effects model to predict cyclist exposure were temperature and concentrations of CO ($R^2 = 0.60$) and NO_2 ($R^2 = 0.74$) measured at urban background and roadside stations, respectively. Asmi et al. (2009) found that the ratio of UFP concentration in the driver's cabin of buses to that measured at an urban background site varied from 1.2 to 6.9 and was dependent on the age of the bus, time of day and route. Zuurbier et al. (2010) systematically evaluated the relationship between bus, car and bicycle UFP exposures and urban background concentrations in Arnhem, the Netherlands. They reported median mode to background ratios of 1.6 (diesel car, electric bus) to 2.5 (diesel bus) and correlations between 0.01 (diesel bus) and 0.87 (bicycle on low-traffic route).

The limited data available to-date indicates that fixed-site monitors may offer some ability to estimate UFP exposure of commuters in areas less affected by vehicle emissions, such as those using subways or low-traffic bike paths. However, depending on location, such persons are likely to constitute only a minor proportion of the commuting population. In the absence of widespread UFP monitoring networks, the utility of routinely monitored particle and gaseous pollutants or individual UFP monitors to represent in-transit UFP exposure appears significantly constrained (Krausse and Mardaljevic, 2005; Vinzents et al., 2005).

8. HEALTH EFFECTS OF IN-TRANSIT UFP EXPOSURE

Studies of health effects due to commuter UFP exposure are summarised in Table S9 in the Supplementary Information file.

8.1 Healthy Individuals

Nystrom et al. (2010) showed that while a cellular response in the airway epithelium was not elicited, minor biological responses such as increased systemic markers of inflammation and signs of lower airway irritation were observed in 20 healthy individuals exposed to subway air (mean UFP concentration 1.1×10^4 particles cm^{-3}) for 2 hours while alternating between exercising on a bicycle ergometer and resting. However, road tunnel air (median UFP concentration 1.1×10^5 particles cm^{-3}) elicited an inflammatory response in the lower airways and elevated levels of T-lymphocytes and alveolar macrophages in brochoalveolar lavages from 16 healthy individuals who followed the same protocol (Larsson et al., 2007). The particle mass concentrations that subjects were exposed to in the two above studies were similar, while UFP and NO_X concentrations were an order of magnitude higher in the road tunnel study than in the subway study due to the presence of proximate vehicle emissions. Although it is not possible to ascribe the disparity in the results of the two studies to differences in UFP concentration alone, the results are suggestive of a causative role for UFP and NO_X in airway inflammation observed following exposure to vehicle emissions.

Thirty-eight healthy volunteers who cycled parallel to a major traffic corridor for 20 minutes (mean UFP concentration 2.9×10^4 particles cm^{-3}) experienced a minor increase in blood inflammatory cell distribution compared to cycling in a clean air environment, although the role of UFPs as distinct from $PM_{2.5}$ was not clear (Jacobs et al., 2010). UFP and EC exposure in 12 healthy non-smoking individuals cycling in traffic (mean UFP concentration 2.8 to 4.1 $\times 10^4$ particles cm^{-3}) for 1 hour was weakly associated with acute effects; decreased lung function and increased exhaled NO (as a marker of airway inflammation) were observed 6 hours post-exposure (Strak et al., 2010). Oxidative DNA damage observed in 15 healthy subjects was positively correlated with cumulative UFP exposure, to which 1.5 hours of cycling during rush hours (mean UFP concentration 3.2×10^4 particles cm^3) contributed substantially and resulted in greater damage compared to indoor cycling on an ergometer (Vinzents et al., 2005). Concentrations of other pollutants (PM_{10}, NO_X, CO) measured at fixed-sites were not associated with oxidative DNA damage.

UFP exposure resulted in modest effects among 34 healthy subjects that commuted by automobile, bus or bicycle for 2 hours (median UFP concentration 2.7 to 4.4×10^4 particles cm^{-3}); peak expiratory flow decreased slightly and airway resistance increased immediately following exposure, and a significant increase in exhaled NO was observed 6 hours post-exposure for automobile and bus commuters, but not cyclists (Zuurbier et al., 2011).

As the respiratory minute ventilation of cyclists is 2 to 4.5 times that of automobile and bus passengers (Zuurbier et al., 2009; Int Panis et al., 2010), the potential dose of inhaled UFPs received during active transport may be significantly higher than that in non-active modes, and recent health effects studies have already begun to adopt a more dose-oriented approach to reflect this (Zuurbier et al., 2011).

8.2 Health-compromised Individuals

Asthmatics: Asthma exacerbations can be triggered due to oxidative stress and inflammation caused by UFPs in the lungs of susceptible individuals (Weichenthal et al., 2007). Reductions in lung function and increased daily symptoms in asthmatics and COPD patients attributable to elevated UFP concentrations have been observed in epidemiologic studies, with more immediate effects seen first in the respiratory system, and a delayed response of cardiovascular effects (Wichmann et al., 2000; Ibald-Mulli et al., 2002).

Consistent asymptomatic reductions in lung function (FEV_1, FVC) and increases in both inflammatory biomarkers and airway acidification were observed in 60 persons with mild or moderate asthma who walked for 2 hours along a busy London street affected by diesel exhaust (median UFP concentration 6.4×10^4 particles cm^{-3}) (McCreanor et al., 2007). The effects were more frequently associated with UFP and EC concentrations than those of $PM_{2.5}$ and NO_2. Significantly reduced respiratory effects were observed when subjects walked along a route less affected by traffic emissions (median UFP concentration 1.8×10^4 particles cm^{-3}).

Fourteen mild asthmatics exposed to road tunnel air (median UFP concentration 2.3×10^5 particles cm^{-3}) for 2 hours while alternating between exercising on a bicycle ergometer and resting experienced no changes in bronchial responsiveness and most lung function parameters, although peak expiratory flow decreased, and minor indications of inflammation were measured in nasal lavages, but not blood samples (Larsson et al., 2010).

Diabetics: Exposure to pollutants (median UFP concentration 4.3×10^4 particles cm^{-3}) during 1.5 to 1.8 hour automobile highway trips made by 21 type 2 diabetics was shown to elicit a decrease in high-frequency heart rate variability the day after exposure, which was more associated with the interquartile range of UFP concentration compared to those of $PM_{2.5}$, NO_2 and CO, albeit not significantly (Laumbach et al., 2010). An increased low frequency to high frequency heart-rate variability ratio was observed immediately post-exposure that was not consistent with other observations, although confounding effects not present in the aforementioned finding may have influenced this result.

Elderly Persons: Nineteen elderly subjects that were exposed to unfiltered and filtered air during 2 hour automobile trips on Los Angeles freeways (mean unfiltered UFP concentration 0.78 to 1.1×10^5 particles cm^{-3}) experienced a 20% decrease in the incidence of atrial ectopic heartbeats and 30% decrease in cardiopulmonary stress biomarkers under the filtered compared to the unfiltered condition (Cascio et al., 2009; Hinds et al., 2010). Other measured parameters (lung function, indicators of inflammation, blood pressure) did not vary significantly between the two conditions. The observed atrial arrhythmia was ascribed to increased intra-atrial pressure, and was associated with UFP concentrations rather than gases or particle mass (Cascio et al., 2009; Hinds et al., 2010). The significance of such events is related to their role in causing more sustained arrhythmias.

8.3 Summary

Commute-time exposure to traffic and attendant pollutant emissions, noise and stress has been associated with increased risk of serious adverse health effects such as myocardial infarction (Peters et al., 2004). The specific role of UFPs as a causative agent of such effects is not clear, and the findings of the limited number of health effects studies addressing commuter exposure to vehicle emissions are mixed. However, some initial trends are emerging. While it is inherently difficult to separate the effects of UFPs from those other pollutants within the real-world exposure scenarios employed by the studies described above, the observed health effects were generally associated most strongly with UFP concentrations. Furthermore, the use of filtered air exposure scenarios in the Los Angeles freeway study (Cascio et al., 2009; Hinds et al., 2010) reduced particle concentration by >95% compared to the unfiltered condition but did not affect the level of gaseous pollutants, yet there was a marked difference in the cardiac effects observed between the two scenarios. The effects observed by McCreanor et al. (2007) were greater in those with moderate compared to mild asthma, and the degree to which this is true of other susceptible groups (i.e. increasing effects with increasing disease severity) is unclear. The 10 commuter health effects studies performed to-date have yielded valuable information, however, it is clear that further studies are required in order to better elucidate the role of UFPs.

9. MODELLING EXPOSURE

9.1 Approaches employed to-date

The ability to accurately model in-transit UFP exposure concentrations has numerous attractive applications in urban planning, transport and policy development. The majority of published studies that developed models employed a multivariate regression approach that incorporated meteorologic, traffic or other pollutants as independent variables (Krausse and Mardaljevic, 2005; Vinzents et al., 2005; Weichenthal et al., 2009; Boogaard et al., 2009; Kaur and Nieuwenhuijsen, 2009). Given the potential for variability in the strength of associations between the independent variables and measured UFP concentrations discussed in sections 5, the external validity of these models is unknown. However, the models were of the explanatory type, and were developed in order to assess the effect of various parameters on UFP concentration measured in a specific location. Their ability to predict exposure concentrations varied from fair (R^2 = 0.35) to very good (R^2 = 0.74). The influence of mode-dependent parameters like ventilation were either included in a qualitative sense (e.g. ventilation setting or window position) or not included at all. This limitation was raised by both Briggs et al. (2008) and Weichenthal et al. (2008).

Several recent studies (Pui et al., 2008; Xu and Zhu, 2009; Knibbs et al., 2010) have sought to overcome the limitations described above by adopting a more mechanistic, mass-balance modelling approach for automobiles. This has been based on measurements of the effects of cabin ventilation, filtration, particle penetration or deposition on in-cabin concentrations (Qi et al., 2008; Gong et al., 2009; Knibbs et al., 2009a; Xu et al., 2010). These studies have generally shown very good results when validated with experimental data. The main limitation of such approaches is that they require the input of an initial on-road or in-cabin UFP concentration. Therefore, there is a clear need to couple models capable of predicting outdoor or on-road concentrations with those focussed on predicting what proportion of these concentrations reach occupants, and how particle dynamics will affect concentrations through time. Moreover, further refinement of models for predicting exposure in active transport modes will be of significant utility. In summary, there is both substantial need and scope for development of models capable of accurate prediction of UFP exposure concentrations in-transit. .

9.2 Spatial and temporal aspects of exposure

Efforts to improve understanding of the spatial and temporal nature of UFP exposures during transit have benefited greatly from the use of Global Positioning Systems (GPS) and Geographic Information Systems (GIS), usually at the measurement and analytical stages, respectively. Gulliver and Briggs (2005) described the development and use of a GIS-based model for predicting exposure to PM_{10} (particles < 10 µm) during transit, however, the application of spatial technologies to UFPs has to-date been limited to a handful of in-transit studies (Hvidberg, 2006; Thai, 2008; Berghmans et al., 2009; Boogaard et al., 2009; Pattinson, 2009; Int-Panis et al., 2010). Synchronised video recordings have been included in some studies (Kaur et al., 2006; Berghmans et al., 2009), which affords an additional perspective from which analyses can be performed.

Given the good level of spatial data quality obtainable from even the more basic mobile telephones at present, the integration of such data into exposure studies will assist data interpretation and help to form a more complete and accurate assessment of pollutant exposure and dose for large study populations (Jerrett, 2010). The appropriateness and capability of mobile telephones to record spatial data and photographs during commuting has already been established by Pooley et al. (2010), and Pattinson (2009) collected such data in addition to UFP measurements when commuting by bicycle.

Land use regression (LUR) is an application of GIS that is gaining momentum as a tool with which to predict exposure to a variety of pollutants (see Hoek et al., 2008). The utility of LUR techniques to predict UFP concentrations and spatial variability is not well-established due to absence of extensive UFP monitoring networks; other (mainly gaseous) pollutants have been the focus of most work performed to-date. However, a recent study has reported reasonable performance of LUR when applied to UFP concentrations in Amsterdam, and comparable predictive utility was observed between the LUR model for UFPs and those for other pollutants (Hoek et al., 2011). LUR is an emerging technology that will increasingly find applications in prediction of personal exposure to a range of pollutants, albeit with an attendant need for validation based on measurements (Hoek et al., 2008). This highlights the need for high-quality databases of concomitant in-transit UFP and spatial measurements.

10. FURTHER RESEARCH NEEDS

10.1 In-transit contribution to daily exposure

The significance of in-transit UFP exposure is highly dependent on personal, demographic and occupational context. UFP concentrations encountered on the commute to and from work will exert much greater influence on the total daily exposure of a non-smoking office worker than a smoker or someone who experiences high occupational exposure. Likewise, the health effects of the same exposure on an adult and child are likely to vary. Without better understanding of the characteristics of 24 hour UFP exposure for numerous demographic groups, knowledge of in-transit exposure alone is of reduced utility. However, it is useful to be able to determine, for a given location, the transport mode in which highest concentrations occur and the factors that determine this. Such information has numerous valuable planning and policy applications.

A handful of studies have estimated the influence of measured in-automobile UFP concentrations on total exposure. Two were based on Los Angeles residents (Zhu et al., 2007; Fruin et al., 2008), and their estimates ranged from 10 to 50% and 33 to 45%, respectively. Wallace and Ott (2011) measured UFP concentrations in a wide range of microenvironments in two US cities and estimated the in-automobile contribution to total exposure to be 17%, which they attributed to the relatively low density of traffic and diesel trucks on the roadways they measured compared to LA. In all cases, the time spent in automobiles was assumed to be about 90 minutes per day. The applicability of the estimates reached by these studies to other regions is unknown, but they have established a range within which automobile commutes in urban areas may be expected to contribute to daily UFP exposure. These estimates have flagged this topic as one requiring further investigation, preferably including several transport modes.

It is important to consider the distinction between UFP concentration and exposure (Krausse and Mardaljevic, 2005). A high concentration experienced for a brief duration can result in a lower exposure than a low concentration for a longer period. This underscores the need for both accurate time-activity pattern data across broad demographic groups and representative UFP measurements within the various microenvironments in which time is spent. Until more expansive UFP exposure studies that follow large groups of people of varying time-activity patterns are completed, the ability to discern the range of commute-time's specific contribution to total exposure is constrained.

10.2 High exposure professions

The magnitude of UFP exposures incurred by people whose occupation requires them to spend extended period in-transit is poorly understood. Professional drivers, bicycle couriers, police officers and other groups whose work day is constituted by long periods in transport microenvironments may all be at risk of substantially elevated exposure compared to the general population. Riediker et al. (2004) reported the negative health effects of in-vehicle $PM_{2.5}$ exposure on young and healthy police officers during 9 hour shifts in patrol cars. Similar studies focussed towards UFPs are required.

10.3 Exposure-health effects link

Various acute human health effects caused by UFP exposures have been investigated in controlled exposure studies using a range of subject groups. However, their relevance to in-transit exposures is unclear. There have been precious few studies that measured the effects of in-transit exposures on health end points, and these were described in section 8. There is a significant need for further studies in this area, as they will serve to bolster the link between exposure and health effects, and this will have implications across policy, planning and public health arenas (de Nazelle and Nieuwenhuijsen, 2010). Furthermore, given the substantial variability in minute ventilation between occupants of different modes (Zuurbier et al., 2009; Int-Panis et al., 2010), the transition from an exposure to dose-oriented approach is likely to yield data of greater relevance to studies of health effects.

10.4 Data from the developing world

A striking feature of the English language literature we searched is the almost complete absence of studies performed in developing regions; with the exception of only the cycling study performed in Bogota, Columbia by Fanara (2003) and cited by Kaur et al. (2007), no other studies from developing countries were identified. This shortcoming is compounded by the generally poor air quality experienced in these regions (Han and Naeher, 2006) and their large populations and urban density. The effect of this combination of factors is that very high UFP exposures are likely to occur for large numbers of people, but the magnitude of such exposures is unknown. Studies of commuter exposure to particulate mass (RSP, PM_{10}) performed in Delhi and Hanoi have reported exceptionally high concentrations (Saksena et al., 2007; Saksena et al., 2008). Moreover, in addition to walking, the most popular modes of transport, such as bicycles, scooters, motorcycles and 3-wheelers (tuk-tuks, auto-rickshaw etc), are unlikely to afford significant protection from the emissions of proximate traffic, which can include a substantial proportion of high emitting two-stroke vehicles. There is a clear need to redress the scarcity of research in this area.

10.5 Other needs

Major needs in future in-transit UFP exposure studies have been outlined above, and numerous other aspects requiring additional research have been suggested throughout this review. Further investigation of the variability inherent in the determinants of exposure discussed in section 5 is required to permit better appreciation of their effects. There is also an obvious need for improved modelling techniques, incorporating GIS, and for further comprehensive assessments of the health risk-benefit balance for active transport modes (de Nazelle et al., 2009; de Hartog et al., 2010).

11. CONCLUSIONS

In our analysis of 47 studies comprising approximately 3000 trips undertaken in 6 transport modes, we found that highest trip-weighted mean concentration occurred in automobile cabins during tunnel travel (3.0×10^5 particles cm^{-3}), and the lowest whilst cycling (3.4×10^4 particles cm^{-3}). Mean concentrations in bus, automobile (non-tunnel travel), rail, and walk modes were generally comparable. However, UFP exposure (and dose) during time spent in-transit is strongly dependent on a range of mode-specific and more general determinants, including, but not limited to, the effects of: meteorology, traffic parameters, cabin ventilation, filtration, deposition, UFP penetration, fuel type, exhaust treatment technologies, respiratory minute ventilation, route and microscale phenomena. Therefore, direct comparison of concentrations measured in different modes highlights general trends, but should not be extrapolated without detailed consideration of the above factors. Characterising the variability in the effects of these determinants will be an important aspect of future work.

There is preliminary evidence to suggest that time spent in-transit can contribute substantially to total daily exposure, and future studies require comprehensive assessment of 24 hour UFP exposures across a broad demographic spectrum. Moreover, the range and variability of acute health effect associated with in-transit exposures are not well understood, and further studies are required to supplement the findings of the limited number performed to-date.

Transport is a ubiquitous component of life, and initial evidence suggests that UFP exposures incurred during this time can contribute substantially to daily exposure and be associated with adverse health effects in susceptible and healthy persons. Further research to better define this link is therefore well-justified, and will be of considerable benefit to urban planning, policy development and public health.

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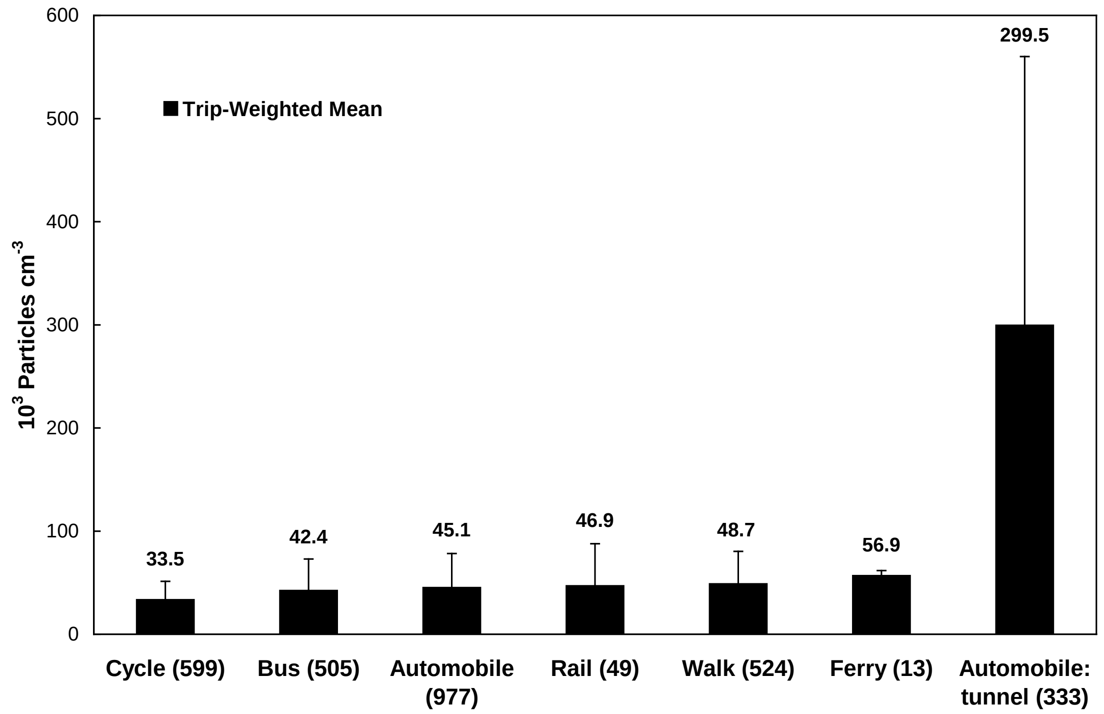

Figure 1. Trip-weighted mean UFP concentrations in each transport mode, shown as bold numbers. The number of trips taken in each mode is shown in brackets. Error bars denote the trip-weighted standard deviation. The studies from which the data were extracted are listed in the Supplementary Information file.

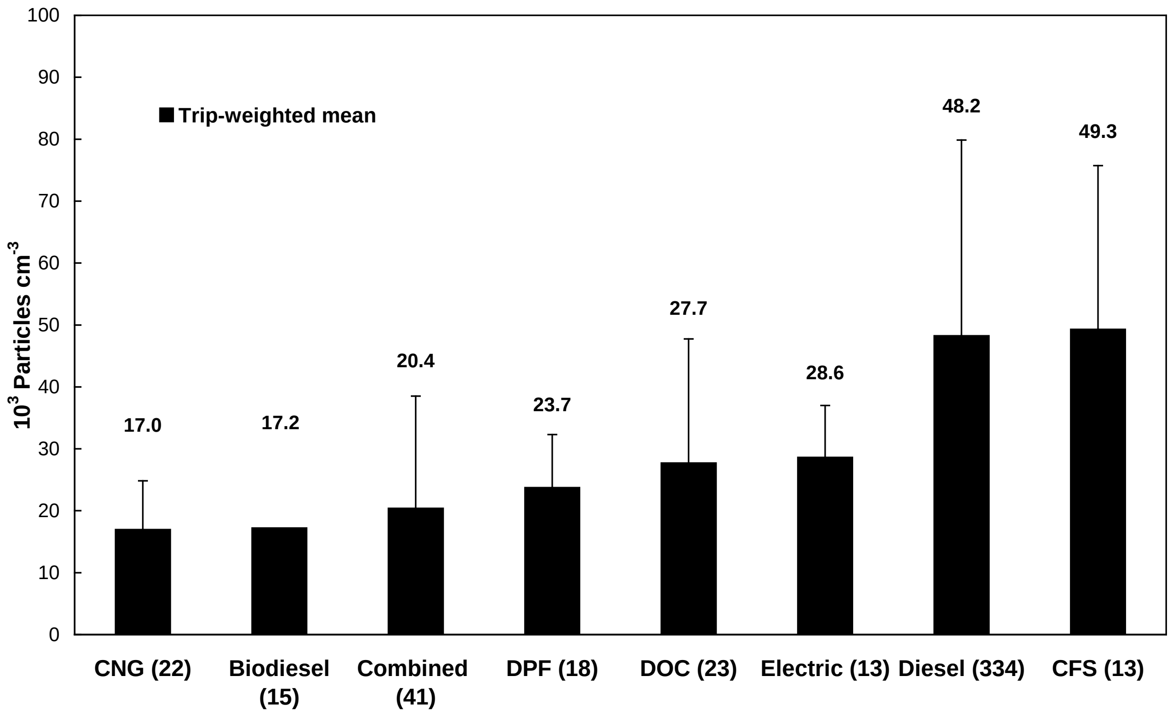

Figure 2. Trip-weighted mean UFP concentrations measured in buses of different fuel type and emission control device. The number of trips taken in each category is shown in brackets. Error bars denote the trip-weighted standard deviation. See text for abbreviations.