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Review article

Particle number size distributions and concentrations in transportation environments: a review

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ABSTRACT

Ambient air ultrafine particles (UFP, particles with a diameter <100 nm) have gained significant attention in World Health Organization (WHO) air quality guidelines and European legislation. This review explores UFP concentrations and particle number size distributions (PNC-PNSD) in various transportation hotspots, including road traffic, airports, harbors, trains, and urban commuting modes (walking, cycling, bus, tram, and subway). The results highlight the lack of information on personal exposure at harbors and railway stations, inside airplanes and trains, and during various other commuting modes. The different lower particle size limits of the reviewed measurements complicate direct comparisons between them. Emphasizing the use of instruments with detection limits ≤10 nm, this review underscores the necessity of following standardized UFP measurement protocols.

Road traffic sites are shown to exhibit the highest PNC within cities, with PNC and PNSD in commuting modes driven by the proximity to road traffic and weather conditions. In closed environments, such as cars, buses, and trams, increased external air infiltration for ventilation correlates with elevated PNC and a shift in PNSD toward smaller diameters. Airports exhibit particularly elevated PNCs near runways, raising potential concerns about occupational exposure. Recommendations from this study include maintaining a substantial distance between road traffic and other commuting modes, integrating air filtration into ventilation systems, implementing low-emission zones, and advocating for a general reduction in road traffic to minimize daily UFP exposure. Our findings provide important insights for policy assessments and underscore the need for additional research to address current knowledge gaps.

1. Introduction

In aerosol science, ultrafine particles (UFP) are conventionally defined as airborne particulates with an aerodynamic diameter smaller than 100 nm. UFP in the atmosphere can have both natural and anthropogenic sources, and can be both primary and secondary. They include particulates that are intentionally produced (engineered), unknowingly generated, emitted from both natural and anthropogenic primary sources, and secondarily formed from precursors during specific atmospheric processes, anthropogenic, and biogenic precursors (Ramírez et al., 2020; Kumar et al., 2014; Morawska et al., 2008). The chemical composition of UFP depends on their source, as well as the regional geography and climate. They comprise elemental and organic carbon (EC and OC), inorganic components (sulfate, nitrate, ammonium, chloride, metals, and metalloids), secondary organic compounds from the oxidation of volatile organic compounds (VOCs), and biological components (microbes, spores, pollens, and biologic debris) (Moreno-Ríos et al., 2022; Ehn et al., 2014). In urban areas, UFP are mostly directly emitted from combustion processes, primarily road traffic, with contributions from airports, ports, and domestic emissions. They can also be newly formed in the atmosphere from precursor gases, such as H_2SO_4, NH_3, and some VOCs (Hopke et al., 2022; Rönkkö et al., 2017; de Foy and Schauer, 2015; Ehn et al., 2014; Kulmala et al., 2012; Seigneur, 2009; Stanier et al., 2004).

Due to their nanometric size and aspects of their behavior, similar to gases, monitoring UFP is challenging (Baldauf et al., 2016). Atmospheric particulates are commonly monitored and regulated in terms of mass per volume of air ($\mu g/m^3$) of PM_{10} and $PM_{2.5}$ (particulate matter with aerodynamic diameters less than 10 μm and 2.5 μm, respectively) for air quality purposes; however, no official air quality standards have been set in place for UFP monitoring, which is conducted in terms of the number of particles per volume of air ($\#/cm^3$). Mounting evidence

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suggests that monitoring $PM_{2.5}$ does not necessarily capture the variability of UFP, probably due to contributions from different sources and the spatial variability of these parameters (Boccuni et al., 2021; de Jesus et al., 2019; Hofman et al., 2016; Eeftens et al., 2015; Wang et al., 2010; Gomišček et al., 2004; Ruuskanen et al., 2001). Thus, although UFP are generally negligible in terms of mass (usually <5 % of $PM_{2.5}$), they dominate the particle number concentration (PNC) (80–90 %) (Trechera et al., 2023; Baldauf et al., 2016; Morawska et al., 1999). The lack of correlation between PM and PNC implies that measures implemented to mitigate PM_{10} and $PM_{2.5}$ may not necessarily lead to a reduction in UFP (de Jesus et al., 2019).

In recent decades, there has been growing awareness of the health effects of exposure to UFP due to their penetration capacity within the respiratory system (EFCA, 2019). When inhaled, UFP can reach the pulmonary alveoli, where a fraction of these particles can penetrate the alveolar-capillary barrier, enter the bloodstream, and circulate throughout the body (Miller et al., 2017; Terzano et al., 2010). Toxicological studies have shown that the toxicity per unit of mass of UFP increases with decreasing particle diameter (Meng et al., 2013; Cassee et al., 2002; Donaldson et al., 2002; Donaldson et al., 1998). Such particles have a very high specific surface area per unit of bulk volume (cm^2/cm^3) and high surface reactivity, and therefore tend to adsorb toxic species (polycyclic aromatic hydrocarbons and heavy metals, among others) and convey them within the body, causing oxidative stress (Kwon et al., 2020). Exposure to ambient UFP can induce numerous issues in the respiratory system, including changes in lung functions, coughing, reduced expiratory flow (Peters et al., 1997), airway inflammation, and enhanced allergic responses (Li et al., 2016; Alberg et al., 2014). UFP have also been associated with cardiovascular issues, such as prothrombotic effects, altered heart rate and heart rate variability, accelerated atherosclerosis, and altered endothelial function (Schraufnagel, 2020; Schulz et al., 2005; Suwa et al., 2002). Markers of brain inflammation and neurodegenerative diseases have also been attributed to UFP exposure (Calderón-Garcidueñas and Ayala, 2022). Finally, digestive problems and a higher incidence of diabetes have also been observed (Bai et al., 2018). The range of health impacts of UFP remain under investigation, with a particular focus on differentiating and comparing the specific health outcomes of UFP and $PM_{2.5}$. For instance, $PM_{2.5}$ has been noted to have more immediate effects, whereas UFP result in more delayed effects but have a greater influence on mortality (Ibald-Mulli et al., 2002).

No consistent evidence exists for independent adverse effects arising from UFP (EFCA, 2019). Consequently, the World Health Organization (WHO) Guideline Development Group (GDG) has stated that the body of epidemiological evidence is not yet sufficient to formulate specific Air Quality Guidelines (AQGs) for UFP (WHO, 2021). In response, the GDG has formulated good practice statements for UFP, addressing concerns about their health and environmental effects. The WHO statements include: 1) the quantification of ambient UFP in terms of PNC, 2) the integration of UFP monitoring into existing air quality monitoring systems, 3) the distinction between low (<1,000 #/cm^3, 24-h mean) and high (>10,000 #/cm^3, 24-h mean; >20,000 #/cm^3, 1-h mean) PNC, and 4) the utilization of emerging technologies to assess the exposure of the population to UFP for epidemiological studies. In view of the WHO (2021) recommendations, the draft of the new European Air Quality Directive (EC, 2023) includes requirements to monitor ambient air concentrations of UFP and particle number size distribution (PNSD) in urban supersites, as well as UFP concentrations in rural areas and pollution hotspots, including traffic, harbors, airports, and industry, among others.

The lack of reference instrumentation and methods to measure PNC and PNSD hindered the monitoring and control of UFP until ACTRIS (2021) and CEN (2016, 2018, 2020, 2023), which led to the implementation of various protocols in many exposure studies. As reported by Trechera et al. (2023) and Rivas et al. (2020), UFP datasets in Europe use different lower size detection limits (between 3 and 20 nm). Because UFP concentrations in this low size range can be significant, comparisons among UFP concentrations reported must be made with caution. Furthermore, although the highest UFP exposures are reported in urban environments (EFCA, 2019), these regions exhibit very high spatial variability in concentrations (Gani et al., 2021). Although road traffic has been identified as a major source of urban exposure to UFP, the contribution from atmospheric nucleation from gaseous precursors can also be significant in specific cities (Hopke et al., 2022, and references therein). Both instrumental and spatial variability, as well as differences in source origin, have likely contributed to inconsistency in previous UFP health studies, as evidenced by EFCA (2019) and WHO (2021).

Several studies have shown that populations are exposed to UFP peaks throughout the day, especially during commuting (pedestrian, bicycle, car, taxi, bus, tram, or metro) or long-distance journeys (e.g., by aircraft, train, and ship) due to the proximity to major air pollutant sources throughout these periods (Moreno et al., 2020; Stacey, 2019, among others). However, although reviews have been conducted to understand UFP in outdoor and indoor urban environments (Hopke et al., 2022; Rivas et al., 2020; Kumar et al., 2014; Presto et al., 2021; EFCA, 2019), aviation (Riley et al., 2021; Frenzel and Kohnert, 2020; Stacey, 2019), highways (Lv et al., 2020), commuting (Rafiepourgatabi et al., 2021; Karanasiou et al., 2014; Knibbs et al., 2011), and road traffic (Moreno-Ríos et al., 2022; Yadav et al., 2019; Morawska et al., 2018), there is still a need for a review to provide a global comparison of UFP and PNC-PNSD in all transportation environments. To this end, the present study reviews the results of studies concerning PNC-PNSD in transportation hotspots, including road traffic, airports, harbors, trains, and other urban commuting modes. The aim of this study is to gain a better understanding of UFP concentrations and PNSD in these exposure hotspots and to obtain information on concentration ranges and particle size patterns. Additionally, this study aims to identify gaps in scientific knowledge to guide future research that may inform policy decisions.

2. Methods

The literature used in this review was obtained by searching the Scopus database for the following terms or combinations of terms: “exposure”, “ultrafine particles”, “nanoparticles”, “particle number concentration”, “size distribution”, “commuting”, “aviation”, “subway”, “shipping”, “road traffic”, “bicycle”, “bus”, “train”, “motorbike”, “scooter”, and “transport”. The Scopus strings used for the search are provided in the supplementary material section (Table S1). Literature searches were limited to July 2022; articles published after this time were not included. Although the use of systematic reviews for environmental and human health risk assessment is encouraged, due to the scarcity of articles meeting the requirements of measuring UFP and PNSD in transport environments, we believe that our review takes into consideration the key papers.

Our original search resulted in the identification of 1046 publications after removing duplicates. An initial selection was made based on the title and subsequently on the abstract of these studies, excluding a total of 863 studies. The remaining 183 studies were subjected to a more in-depth analysis, focusing on studies reporting measurements of ambient air or personal air monitoring in transport hotspots. This further screening process excluded any studies that i) did not report complete information for both PNC and PNSD measurements, ii) were based on modeling without experimental data, iii) were conducted at urban or regional background monitoring stations, and iv) were performed indoors or at the laboratory scale. Additionally, studies for which full-texts were not available were also excluded, resulting in a final selection of 49 studies. The selection process is summarized in Fig. 1.

As stated above, UFP dominates the PNC (80–90 %) (Trechera et al., 2023; Baldauf et al., 2016; Morawska et al., 1999). Accordingly, good practice statements on UFP (WHO, 2021; EC, 2023) recommend quantifying ambient UFP in terms of PNC for a particle size range ≥10 nm, because of the high uncertainties associated with measuring <10 nm. No

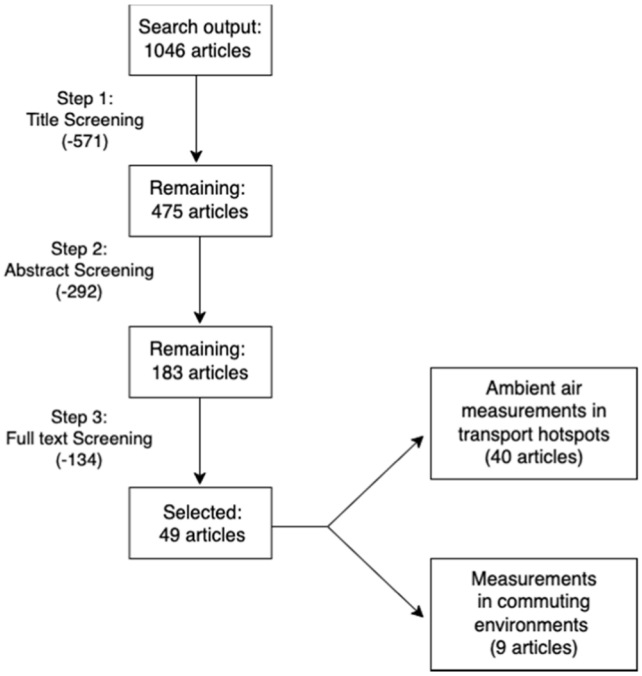

Fig. 1. Flow diagram outlining the literature review process.

restriction is placed on the upper limit because, in urban areas, PNC from 100 to 800 nm accounts for only 10–20 % of the total PNC (consequently, UFP and PNC are used as surrogates in EC, 2023). Therefore, in this review, UFP and PNC are used interchangeably. For all studies, information on site location, measurement duration, season, instrumentation deployed, particle size range covered, PNC, major size mode, and second and third modes was collected and reported. Furthermore, when available, data for average particle size (nm), $PNC_{<25nm}$, $PNC_{25–100nm}$, $PNC_{>100nm}$, NO_X, BC, PM_{10}, $PM_{2.5}$, PM_1, SO_2, and CO were reported. The urban background PNC levels of the corresponding cities were included for comparison purposes when reported. The selected literature was then sorted into the following categories: PNC and PNSD (i) in road traffic environments, (ii) on airport grounds and near airports, and (iii) for different commuting modes (pedestrian, bicycle, subway, bus, trams, and in-vehicle). The main results reported in the selected studies are summarized in Tables 1–3.

As described later, a disparate range of instruments were used for measurements in the studies included in this review. Tables 1–3 show the instruments and the specific particle sizes covered in each case; however, it is beyond the scope of this study to discuss the different instruments available for measuring PNC and PNSD. Readers are referred to Kumar et al (2010) and Kangasluoma et al. (2020) for detailed information on the instruments used and the differences between them.

3. Results

3.1. UFP in road traffic

Road traffic is considered the main source of UFP in urban environments (Hopke et al., 2022). Our review identified 31 eligible articles (Table 1) containing complete information about PNC and PNSD near roads. Most of these studies were carried out in Europe (61 %), followed by America (23 %), Asia (13 %) and Oceania (3 %). Mobility particle sizers, including the Scanning Mobility Particle Sizer (SMPS), Differential Mobility Spectrometer (DMS), Differential Mobility Particles Sizer (DMPS), and Mobility Particle Size Spectrometer (MPSS), were used in 19 studies (61.3 %). The Fast Mobility Particle Sizer (FMPS) was employed in five studies, the Electrical Low Pressure Impactor (ELPI+) in three studies, the Engine Exhaust Particle Sizer (EEPS) in two studies, and the Portable Aerosol Mobility Spectrometer (PAMS) and UFP TOPAS in one study each. Ten of the selected studies were long-term studies (>6 months) including both cold and warm season measurements, seven studies were conducted between 1 and 6 months, nine studies for less than 1 month, and six studies over short-term period (<1 week). Out of the 31 studies considered, 21 (68 %) included night measurements, whereas the remaining ten studies only included daylight hours measurements. Traffic volumes were reported by 20 studies (65 %) and the corresponding and simultaneous measurements of urban background concentrations were reported by 16 studies (52 %). The most common particle size range measured was from 4 or 5 nm up to 560 or 800 nm, with a few exceptions starting measurements from 20 nm. For comparison, we distinguished between studies conducted over longer durations, encompassing both day and night measurements, and shorter-term studies often limited to the daytime hours or a few days.

Considering only studies including both day and night measurements (which may better represent potential exposure, Pelliccioni et al., 2020), carried out for more than one month (15 studies), the average PNC was found to be 25,977 ± 20,494 #/cm^3 (p_{25}: 12,996 #/cm^3; p_{50}: 21,801 #/cm^3; p_{75}: 28,825 #/cm^3). From these, a study focusing on the Middle East (Al-Dabbous and Kumar, 2014) recorded the highest concentrations (96,000 ± 60,000 #/cm^3). The minimum concentrations reported were obtained at two relatively small European cities (Strasbourg and Leipzig), recording 8,444 #/cm^3 (Chatain et al., 2021) and 8,617 #/cm^3 (Sun et al., 2019), respectively. In addition to the small size of the cities, the differences compared to the averages of other studies may be due to several factors. Chatain et al. (2021) was the only study considered that used a UFP TOPAS 3031 with a cutoff of 20 nm, thereby losing a significant number of particles in the size range <20 nm.

Indeed, urban background concentrations reported in the same study (6,844 #/cm^3) were not significantly different from those measured at traffic sites. In the case of the low PNC values found in Sun et al. (2019), this may be explained by the use of small cities with low urban background PNC levels (4,902 particles/cm^3). On average, higher PNCs were obtained in Europe (24,986 ± 12,171 #/cm^3) than in the US (16,303 ± 5,374 #/cm^3), probably due to the higher proportion of diesel vehicles in the former.

On the other hand, when considering only studies carried out for more than one month and during daylight hours, the average PNC was 118,252 ± 116,440 #/cm^3. This substantial difference in average concentrations highlights the importance of studying diurnal hourly averages when evaluating the true exposure of citizens to UFPs. Furthermore, having access to year-long representativity is also crucial. The highest PNC values measured ranged from 129,500 #/cm^3 (Fujitani et al., 2012) to a maximum of 360,000 #/cm^3 (Li et al., 2018), all of which were obtained during short-term diurnal measurements. Fujitani et al. (2012) conducted their analyses in Kawasaki, which exhibits an unusually high urban background PNC of 52,800 #/cm^3. Consequently, the increase due to traffic was equal to 2.5, consistent with reports by other selected studies, with a mean increment of 2.8 ± 1.6. On the other hand, the highest PNC values found by Li et al. (2018) can be primarily attributed to the fact that the study was carried out in a road tunnel with limited ventilation and an average traffic volume of 107,000 vehicles per day. PNC <100 nm was only available in ten of the 31 selected studies (32.2 %), indicating that UFP accounted for an average of 87.3 % of the total PNC.

All of the selected studies reported a PNSD peak around 10–30 nm. Unimodal distributions were characteristics of 11 of studies, 14 showed a bimodal size distribution with a second peak at 40–90 nm, and four exhibited a trimodal size distribution. Among the latter, Mishra et al. (2015) was the only study reporting an accumulation mode at 130–180 nm. This difference in size distribution compared to the other studies may be attributed to the fact that it was conducted inside a road tunnel. Studies conducted in Europe exhibited a main mode at 10–30 nm and a secondary mode at 60–90 nm. Studies from Australia, Canada, and the

Table 1
Ambient measurements of PNCs and PNSDs at road transport environments.

Authors	Country	City	Season	Duration	Instrumentation	Day/ Night	Particle size range (nm)	Traffic (veh/day)	PNC (#/cm^3)	PNC Ref (#/cm^3)	Major mode (nm)	Second mode (nm)	Third mode (nm)	Ancillary pollutants
Leavey et al. (2017)	Missouri	St. Luis	c	25 d	P-TRAK UPC 8525 (TSI) & PAMS (Kanomax)	2	20–1000	–	16,819	–	10–30	90–110	/	NO_X, $PM_{2.5}$, CO
Goel and Kumar (2015)	UK	Guildford	c	24 h	DMS50	2	5–560	–	81,000	–	5	10	75	–
Hilker et al. (2021)	Canada	Toronto	w/c	13 y	FMPS 3091 (TSI)	1	10–300	17,200	11,700	–	20	/	/	–
Saha et al. (2019)	Pennsylvania	Pittsburgh	c	6 w	FMPS 3091 (TSI) & CPC 200P	1	5.6–560	12,000	15,000	7,500	< 20	40	/	–
Vouitsis et al. (2015)	Greece	Thessaloniki	w/c	2 m	SMPS 3936L & CPC 3776 (TSI)	1	14–710	–	29,000	7,846	29–51	/	/	NO_2, NO_X, O_3, PM_{10}, $PM_{2.5}$, CO
Joodatnia et al. (2013a)	UK	Guildford	w	6 d	DMS50	2	5–560	–	37,500	–	70	10	6	–
Goel and Kumar (2016)	UK	Guildford	c	126 h	DMS50	2	5–560	64,400	34,400	–	5.6	10	56	–
Fujitani et al. (2012)	Japan	Kawasaki City	w/c	15 d	EEPS 3090 & CPC 3007 (TSI)	2	5.6–560	52,500	129,500	52,800	10–30	40–60	/	PM_{10}, $PM_{2.5}$
Sun et al. (2019)	Germany	Leipzig	w/c	5 y	MPSS	1	5–800	30,666	8,617	4,902	20	/	/	BC
Krecl et al. (2017)	Sweden	Stockholm	w	4 m	DMPS & CPC 3032 (TSI)	1	28–410	30,700	25,000	–	23	/	/	NO_X, BC
Mishra et al. (2015)	Belgium	Antwerp	c	15 d	SMPS 3936 & CPC 3025A (TSI)	2	14—710	120,000	239,100	–	25	43	130–180	–
Pirjola et al. (2012)	Finland	Helsinki	c	2 w	SMPS 3080 & CPC 3025 (TSI)	2	10–420	40,000	51,200	13,700	22	70	/	NO_2, NO_X, BC, $PM_{2.5}$
Sabaliauskas et al. (2012)	Canada	Toronto	w/c	5 y	FMPS 3091 (TSI)	1	5.6–560	20,000	22,210	–	20	/	/	–
Gramsch et al. (2009)	Chile	Santiago de Chile	c	15 d	DMPS	1	10–700	70,000	36,300	9,060	20–30	90	/	PM_{10}, PM_1
Ketzel et al. (2003)	Denmark	Copenaghen	w/c	7 m	DMPS & CPC 3010 (TSI)	1	10–700	26,000	21,392	8,977	20–30	/	/	NO_X, PM_{10}, CO
Zhang et al. (2022)	China	Beijing	c	4 m	SMPS 3938 & CPC 3772 (TSI)	1	11.3–552.3	–	24,868	–	20	70	/	–
Chatain et al. (2021)	France	Strasbourg	c	4 m	UFP TOPAS 3031	1	20–800	28,000	8,444	6,844	< 20	/	/	–
Rivas et al. (2020)	UK	London	w/c	2.5 y	SMPS 3080 & CPC 3022A (TSI)	1	17–640	80,000	12,328	5,630	34	/	/	NO_2, BC, O_3, PM_{10}, $PM_{2.5}$, SO_2, CO
Rivas et al. (2020)	Finland	Helsinki	w/c	2.5 y	DMPS	1	6–800	28,000	15,482	7,003	13.4	/	/	NO_2, BC, O_3, PM_{10}, $PM_{2.5}$, SO_2, CO
Manigrasso et al. (2019)	Italy	Rome	w	3 w	FMPS 3091 (TSI)	1	5.6–560	–	11,905	5,695	7–16	90	/	–
Li et al. (2018)	Pennsylvania	Pittsburgh	c	60 h	SMPS 3081 & CPC 3025 (TSI)	1	4–550	107,000	360,000	–	16	/	/	–
Rahman et al. (2017)	Australia	Brisbane	w/c	3 m	SMPS 3071 & WCPC 3782 (TSI)	1	8–400	12,000	10,500	–	12	50–70	/	NO_X, CO
Wang et al. (2016)	Italy	Piacenza	w/c	8 m	ELPI+ (Dekati)	1	7–10000	40,000	17,200	8,750	20	70	/	$PM_{2.5}$
Al-Dabbous and Kumar (2014)	Kuwait	Fahaheel	w	1 m	DMS500	1	5–1000	–	96,000	–	12	/	/	–
Price et al. (2014)	UK	Swansea	w/c	9 m	ELPI+ (Dekati)	1	7–10000	–	40,562	–	–	–	–	–

(*continued on next page*)

Table 1 (*continued*)

Authors	Country	City	Season	Duration	Instrumentation	Day/ Night	Particle size range (nm)	Traffic (veh/day)	PNC (#/cm^3)	PNC Ref (#/cm^3)	Major mode (nm)	Second mode (nm)	Third mode (nm)	Ancillary pollutants
Birmni et al. (2013)	Germany	Dresden	w/c	6 m	TDMPS (Tropos)	1	5–800	21,000	28,300	5,900	10	90	/	BC, PM_{10}
Buonanno et al. (2011)	Italy	Cassino	w	5 m	FMPS 3091 & CPC 3775 (TSI)	1	5.6–560	17,280	42,000	5,800	30–40	20	/	PM_{10}, $PM_{2.5}$, PM_1
Wang et al. (2008)	Texas	Corpus Christi	w/c	1 w	SMPS 3936 & CPC 3785 (TSI)	2	6–220	–	66,000	–	10–30	50–70	/	–
Roth et al. (2008)	France	Strasbourg	w/c	1 m	ELPI+ (Dekati)	1	7–10000	–	39,000	9,000	7–29	60	/	–
Shi et al. (2007)	China	Beijing	c	1 d	EEPS	2	5.6–560	–	297,800	53,500	10	40	/	–
Shi et al. (2001)	UK	Birmingam	c	4 d	SMPS 3071 & CPC 3022A (TSI)	2	9.6–352	–	95,000	–	20–30	/	/	–

w: warm; c: cold; d: days; h: hours; m: months; y: years; 1: 24 h continuous measurements; 2: daylight hours measurements.

Table 2
Ambient measurements of PNCs and PNSDs near airports.

Authors	Country	City	Refernce site	Season	Duration	Instrumentation	Particle size range (nm)	Time res	Day/ Night	PNC (#/cm^3)	Distance from runway (m)	Major mode (nm)	Second mode (nm)	Ancillary pollutants
Hsu et al. (2013)	California	Los Angeles	NO	w	42 d	FMPS 3091 (TSI)	5.6–560	1 s	1	150,000	–	< 20	/	–
Pirhadi et al. (2020)	Netherlands	Amsterdam	NO	w	32 d	SMPS 3936 & CPC 3788 (TSI)	10–225	3 min	2	35,308	300	20	/	NO_X, BC, $PM_{2.5}$, CO
Psanis et al. (2017)	Greece	Mytilene	NO	w	1 d	SMPS 3034 & CPC 3775 (TSI)	2.5–500	–	2	–	50	20–40	/	–
Buonannoet al. (2012)	Italy	Pomezia	NO	w	1–2 m	FMPS 3091 & CPC 3775 (TSI)	5.6–560	5 min	2	26,000	–	25	/	PM_{10}
Keuken et al. (2015)	Netherlands	Amsterdam	YES	w/c	2 m	SMPS 3034 & CPC 3010 (TSI)	4–487	3 min	2	17,300	7,000	17	/	BC
Tremper et al. (2022)	UK	London	YES	w/c	7 m	SMPS 3080 & CPC 3775 (TSI)	15–661	15 min	1	11,745	1,600	20	–	NO_2, NO_X, BC, PM_{10}
Stacey et al. (2020)	UK	London	YES	w/c	2 m	SMPS 3080 & CPC 3775 (TSI)	15–661	5 min	1	8,911	170	20	/	NO_2, BC, PM_{10}, $PM_{2.5}$
Masiol et al. (2017)	UK	London	YES	w/c	2 m	SMPS 3080 & CPC 3775 (TSI)	14–673	5 min	1	19,000	1,200	14	30–50	BC
Masiol et al. (2016)	Italy	Venice	NO	w	1 m	SMPS 3080 & CPC 3022A (TSI)	14–673	5 min	1	14,000	110	< 14	80	NO_2, NO_X, BC, O_3, $PM_{2.5}$, SO_2

w: warm; c: cold; d: days; h: hours; m: months; y: years; 1: 24 h continuous measurements; 2: daylight hours measurements.

Table 3
Commuters exposure measurements of PNCs and PNSDs at different commuting modes.

Commuting	Authors	Country	City	Season	Duration	Instrumentation	Particle size range (nm)	PNC (#/cm^3)	Average size (nm)	Major mode (nm)	Second mode (nm)
1. Pedestrian	Moreno et al. (2020)	Spain	Barcelona	w	12 h	Hy-SMPS (Hanyang University)	9.5–241.4	–	–	40–70	/
	Ragettli et al. (2013)	Switzerland	Basel	w/c	24 d	miniDiSC	10–300	19,481	46.1–51.2	–	–
2. Bicycle	Moreno et al. (2020)	Spain	Barcelona	w	12 h	Hy-SMPS (Hanyang University)	9.5–241.4	–	–	30–70	/
	Ragettli et al. (2013)	Switzerland	Basel	w/c	24 d	miniDiSC	10–300	22,660	46.1–51.2	–	–
3. Bus	Moreno et al. (2020)	Spain	Barcelona	w	12 h	Hy-SMPS (Hanyang University)	9.5–241.4	–	–	40–70	/
	Ragettli et al. (2013)	Switzerland	Basel	w/c	9 m	miniDiSC	10–300	14,055	46.1–51.2	–	–
4. Tram	Moreno et al. (2020)	Spain	Barcelona	w	12 h	Hy-SMPS (Hanyang University)	9.5–241.5	–	–	50–100	/
	Ragettli et al. (2013)	Switzerland	Basel	w/c	9 m	miniDiSC	10–300	18,818	46.1–51.2	–	–
5. Subway	Moreno et al. (2020)	Spain	Barcelona	w	12 h	Hy-SMPS (Hanyang University)	9.5–241.6	–	–	20	50–80
	Kim et al. (2017)	Korea	Seoul	c	4 d	SMPS 3910 (TSI)	10–420	15,792	–	–	–
	Mendes et al. (2018)	Greece	Athens	w	1 m	SMPS 3080 & CPC 3776 (TSI)	14.6–430	12,000	–	20–30	60–70
6. Passenger cars	Moreno et al. (2020)	Spain	Barcelona	w	12 h	Hy-SMPS (Hanyang University)	9.5–241.4	–	–	40–80	/
	Ragettli et al. (2013)	Switzerland	Basel	w/c	9 m	miniDiSC	10–300	31,784	45–55	–	–
	Leavey et al. (2017)	Missouri	St. Luis	c	25 d	P-TRAK UPC 8525 (TSI) & PAMS (Kanomax)	20–1000	12,975	–	10–30	/
	Goel and Kumar (2015)	UK	Guildford	c	24 h	DMS50	5–560	27,000	–	10	75
	Joodatnia et al. (2013a)	UK	Guildford	w	1 m	DMS50	5–560	27,200	53	10	60
	Zhu et al. (2007)	California	Los Angeles	w	1 d	SMPS 3080 & WCPC 3785 (TSI)	7.9–217	9,100	–	10–30	50–70
	Joodatnia et al. (2013b)	UK	Guildford	c	2 d	DMS50	5–560	57,800	–	10	60

w: warm; c: cold; d: days; h: hours; m: months.

USA exhibited a similar major fine mode at 10–30 nm, with some studies reporting a secondary mode at either 40–70 or 90–110 nm, likely due to the urban background. In Asia and the Middle East, very similar results were obtained, with a major mode at 10–30 nm and a secondary mode at 40–70 nm. The finest major mode occurring at 10–30 nm in most studies can be attributed to i) UFP emissions from gasoline engines (Hopke et al., 2022, and references therein), and/or ii) the nucleation of semi-volatile organic compounds (SVOCs) emitted by diesel engines that escape from the filter traps (Damayanti et al., 2023, and references therein), whereas the modes at 40–90 nm probably arise from primary diesel engine emissions (Hopke et al., 2022, and references therein).

It is important to highlight that relatively few studies report PNC-PNSD datasets with a lower particle size detection limit of at least 10 nm. For these studies, 43–65 % of the PNC falls in the nucleation mode, and 21–53 % and 1–14 % fall within the Aitken and accumulation modes, respectively.

Only six studies reported separate data for warm and cold seasons, and it is therefore difficult to evaluate seasonal variability in the road traffic results. All studies presenting such measurements noted higher PNCs during winter, with increases ranging from 1.34 times higher (Birmni et al., 2013) to 2.8 times higher (Fujitani et al., 2012) than summer PNCs.

3.2. UFP in airports

Aviation-related emissions can have adverse impacts on air quality over large areas surrounding airports (Hudda et al., 2018). Although concerns about UFP from the aviation sector have been growing in recent years, few studies have presented a complete assessment of UFP emissions within or around airport grounds, partly due to logistical and bureaucratic difficulties. Out of all the aviation-related studies analyzed for this review, only nine were selected (Table 2). Eight of these (89 %) were conducted in Europe using different models of SMPSs coupled with CPCs, and an FMPS was used in one case. The remaining study from the US (11 %) was conducted using an FMPS. All of these studies had a duration between 1 and 2 months, although Psanis et al. (2017) only reported results for a few typical days. Five studies (55.5 %) measured a particle size range from 14 or 15 nm up to 700 nm, Pirhadi et al. (2020) measured PNSD from 10 to 225 nm, and the other three studies (33.3 %) measured from 2.5, 4.0 and 5.6 nm up to 500 nm.

Unlike the studies conducted in road traffic sites, which are quite similar except for the type of circulating fleet and climate, aviation-related papers differ significantly. Indeed, five out of the nine studies were performed within airport grounds or just outside airports, three nearby (1–7 km), and one within an aviation base. Moreover, five studies only considered the hours during the highest aircraft activity, and only four studies considered 24-hour observations. Thus, it is difficult to derive average results, and it is instead necessary to evaluate

specific cases.

The highest median PNC of 150,000 #/cm^3 for a 42-day study in summer, including non-activity hours, was found by Hsu et al. (2013) at a monitoring site at the end of the departure runway of LAX airport (1,700–2,200 flights per day). In the same study, monitoring was performed at different distances from the runway, revealing a sharp drop in PNCs at 250 m and 500 m from the runway (19,000 #/cm^3 and 17,000 #/cm^3, respectively), suggesting that the selection of the measurement site inside the airport plays a crucial role in the results obtained. Such high median concentrations could be attributed to the fact that, in this study, an instrument with 1 s time resolution was used to measure from 5.6 nm, with the capability to detect very high concentration peaks (p_{95}: 2,000,000 #/cm^3; p_{99}: 7,100,000 #/cm^3), whereas, in the other studies, the time resolution was approximately 5 min. Psanis et al. (2017) found an increase in PNC from 7,000 #/cm^3 during non-activity hours and up to 800,000 #/cm^3 during hours with high aircraft activity, at 50 m from the runway of a small airport of the Aegean Sea Insular Region (10–15 flights per day) on a typical aircraft activity day.

Among the other selected studies conducted within or just outside airport grounds, those measuring only during aircraft activities determined average PNCs of 30,654 ± 6,581 #/cm^3, whereas those measuring continuously during day and night determined average PNCs of 11,455 ± 3,598 #/cm^3. PNCs measured outside airports at distances from 1.2 to 7 km were, on average, 16,015 ± 3,794 #/cm^3. Among these studies, no notable differences were found to distinguish between 24-hour continuous measurements and measurements during airport activities only.

Regarding PNSD, more homogeneous results were found in the selected studies: seven out of nine studies (77.8 %) reported a unimodal PNSD with a major mode <20 nm, whereas two out of nine studies (22.2 %) reported a bimodal PNSD with a major mode <14 nm and a second mode at either 30–50 (Masiol et al., 2017) or 80 nm (Masiol et al., 2016). These latter modes coincide with those reported for road traffic (Hopke et al., 2022). Keuken et al. (2015) and Buonanno et al. (2012) found that PNSD shifted to larger particles (20–40 nm and 80 nm, respectively) when measuring at background sites and sites downwind from airports, although other studies report similar modes for road traffic (Hopke et al., 2022).

Mazaheri et al. (2009) studied aircraft emissions during idling, taxiing, and landing, showing that PNC increased in the order landing > taking off, with concentrations reaching 1,500,000–2,000,000 #/cm^3 around 50 m from the turbines. The modes of the PNSD at this distance reached 12–22 nm. As the studies cited above show, PNC close to emission points lead to very high daily medians that reach 150,000 #/cm^3 close to the departure runway; however, due to the high atmospheric dispersion, these decrease to 19,000 #/cm^3 at 250 m from the runway. At this distance, the contribution of road traffic emissions to the PNC is likely also important in an airport setting. The larger relative contribution of road traffic PNC as distance from the departure runway increases should be considered when evaluating the impact of UFP from airports in cities. Thus, close to the departure runway, the dominant mode of the PNSD is 12–15 nm. The atmospheric transport of such emissions into cities might lead to an increase in this mode, to 20–25 nm, driven by particle growth and agglomeration. However, airports also suffer from high road traffic emissions, with major modes at 25–30 nm and 70–100 nm. The mixture of these emissions with those from the aircraft during the transport of the pollution plume toward the city might also result in a mode at 20–25 nm. Thus, the impact of the airport's pollution plume on PNCs in cities includes both the emissions from the aircraft and the associated road traffic.

The observed variability among the PNC results underscores the complexity of their sources and emphasizes the central role played by meteorological conditions. Airports are often located in very windy areas, facilitating the rapid dispersion of pollutants, which results in a decrease in PNCs. On the other hand, the high presence of VOCs and SVOCs precursors emitted by jet engines, combined with the clean atmosphere resulting from pollutant dispersion, favors the nucleation of new particles. These particles can grow and/or agglomerate very rapidly, yielding particles in the 10–15 nm range.

When considering the characteristics of the source, it is crucial that studies are carried out using instruments with: i) the lowest possible time resolution in order to capture the PNC peaks before pollutants are dispersed as a function of the meteorological conditions, and ii) the capability to measure particles starting from the smallest possible diameter to capture nucleation mode particles.

Among the studies reviewed, only Masiol et al. (2017) provided separate data for warm and cold seasons. They observed slightly higher PNCs during the cold season (22,000 vs. 19,000 #/cm^3), with a slight shift toward larger particles in the PNSD (first peak: 15–24 vs. 14 nm; second peak: 70–100 vs. 30–50 nm for cold and warm seasons, respectively).

3.3. Commuting

3.3.1. Pedestrian, bicycle, bus & trams

Assessing human exposure to urban UFPs remains a challenge due to the complexity and cost of instruments required for such studies (Sioutas et al., 2005). Conducting studies in equipped air quality monitoring sites and mobile units, while common, may not fully represent realistic personal exposure (Watson et al., 1988; Ott, 1982). Although some portable instruments allow the study of total PNC, assessing PNSD remains a complex challenge for which only a few studies on commuters' exposure are available.

In the reviewed literature, two eligible studies (Table 3) provided information on commuters' exposure to UFPs, both of which were conducted in Europe. Moreno et al. (2020) utilized portable miniaturized scanning mobility particle sizers (Hy-SMPS; Hanyang University; size range: 10–241 nm) to assess UFP exposure in various commuting modes within the city of Barcelona (Spain), during three days in the warm season. Ragettli et al. (2013) employed a miniature Diffusion Size Classifier (MiniDiSC; size range: 10–300 nm) to evaluate commuters' exposure in the city of Basel (Switzerland) over a nine-month study period.

Regarding active commuting, such as walking and cycling, Ragettli et al. (2013) observed slightly higher PNCs for cycling (22,660 ± #/cm^3) compared to walking (19,481 ± 11,705 #/cm^3). However, these differences are likely influenced by the proximity of pedestrian and bike lanes to the road, rather than the mode of active commuting itself. The study also highlighted that UFP exposure during bicycle commutes contributed to more than 20 % of the total daily exposure in winter; this proportion decreased when avoiding main roads. Average particle size, however, showed no significant differences between cycling and walking commutes (42.6 ± 6.5 nm and 43.0 ± 7.9 nm, respectively).

It is important to note that inhalation rates during cycling may be considerably higher than during walking, potentially resulting in an increased UFP dose for the same exposure concentration.

Moreno et al. (2020) also revealed interesting fluctuations in PNC and PNSD based on the route chosen and the time of the day, both for walking and cycling commutes. The PNSD exhibited a unimodal distribution with a peak in the Aitken mode (30–70 nm, conventionally attributed to road traffic emissions). However, Moreno et al. (2020) noted that the PNSD could rapidly shift to finer diameters (peak at 20 nm) both due to increased fresh proximal exhaust emissions and also as a result of bursts of new particle formation (NPF) events in the city.

In the case of buses and trams, the inside air quality is influenced by various factors, including external traffic, the efficiency of the ventilation system, and, in the case of buses, the type of fuel used and subsequent self-pollution. Ragettli et al. (2013) found that the average PNC inside a bus was 14,055 ± 7,951 #/cm^3, making it the least polluted commuting microenvironment in their study. These lower levels could be attributed to the fact that, during the 2010 period, most new buses

recirculated and filtered air mechanically, with many fueled by compressed natural gas (CNG) and diesel (the latter being equipped with filter traps). In the same study, higher PNCs of 18,818 ± 8,120 #/cm^3 were measured inside trams, possibly due to increased ventilation (inflow of outdoor air into the tram, due to partly open windows), an increased number of stops, and the trams' position in the middle of the road. The average particle size ranged between 46 and 51 nm for both buses and trams.

Additionally, Moreno et al. (2020) reported the lowest PNCs inside air-conditioned buses, with size range peaks between 40 and 80 nm. Concentrations inside trams were similar, with the main difference occurring during NPF events, when the PNSD inside the tram shifted toward finer particles, peaking at approximately 20 nm, while that of the bus remained unchanged. In general, enclosed environments such as buses and trams experienced lower PNCs than active commuting modes. However, it is essential to note that increasing the enclosure of buses or trams (by reducing ventilation with outdoor air) raises the risk of transmission of respiratory diseases (Querol et al., 2022).

3.3.2. Subway

Subways serve as widely used public transport systems across the globe, particularly in large metropolises, where they offer advantages in terms of air quality by contributing to the reduction of vehicular road traffic (Martins et al., 2016). However, subways also represent a microenvironment susceptible to poor air quality, and their ventilation systems may not always ensure the provision of sufficiently clean air for commuters (Xu and Hao, 2017). Consequently, conducting studies in different cities is crucial to identify optimal solutions that guarantee healthy air quality inside subways. Despite their importance in this regard, relatively few studies on UFPs have been conducted in subway environments due to the challenges associated with studying these particles.

Three articles meeting the criteria of this review were selected (Table 3) for further analyses. Each of these studies measured particles between 10 and 250 or 400 nm. Two studies used portable instruments in the subways of Barcelona (Moreno et al., 2020) and Seoul (Kim et al., 2017), whereas Mendes et al. (2018) used an SMPS coupled with a CPC to study the subway in Athens. All three studies reported similar results, with an average PNC of 13,896 ± 2,681 #/cm^3. A trimodal PNSD with peaks at 20–30 nm, 40–80 nm and coarser than 100 nm was observed in each study. Simultaneous urban background measurements conducted in these studies suggested that the dominant source of UFP inside the subway was the outdoor urban environment rather than sources directly related to the subway itself. However, the subway did contribute to an increase in the number of particles with diameters >100 nm and to most of the PM_{10} and $PM_{2.5}$ mass (Martins et al., 2016; Xu and Hao, 2017, among others).

3.3.3. Passenger cars

Commuting by passenger car, especially during rush hours, exposes individuals to high air pollutant levels, primarily from the inflow of highly polluted road air and partially from self-pollution. This review identified seven studies (Table 3) reporting PNC and PNSD inside vehicles: five of these (71.5 %) were performed in Europe and two (28.5 %) in America. Distinct instruments were used: three studies (43 %) employed a DMS50 to measure particles from 5 to 560 nm in size, whereas the other four utilized an SMPS (7.9–217 nm), a P-TRACK UFP (20 nm–1 μm), a MiniDiSC (10–300 nm), and a Hy-SMPS (9.5–241.4 nm). The durations of these studies were typically short, spanning a few days to one month, with the exception of Ragettli et al.'s (2013) nine-month study.

The average PNC observed inside vehicles was 27,643 ± 17,240 #/cm^3 (p_{25}: 16,481 #/cm^3; p_{50}: 27,100 #/cm^3; p_{75}: 30,638 #/cm^3). Joodatnia et al. (2013) reported the highest concentrations (57,800 ± 40,600 #/cm^3) in Guildford (UK). The elevated concentrations in this case may be attributed to the intake of outdoor air through the ventilation system. The lowest concentrations (9,100 ± 3,300 #/cm^3) were reported by Zhu et al. (2007) in Los Angeles. This discrepancy could be attributed to the study's location on a highway, providing higher dilution possibilities than urban street canyons. Additionally, the study in Los Angeles implemented measures such as closed windows, a manufactured installed filter, and different ventilation settings, thereby protecting the in-cabin environment from outside air pollution infiltration.

Several studies have indicated that indoor air recirculation and closed windows can lead to significantly lower PNC (reductions of up to 80 %) than well-ventilated interiors during car commuting (Goel and Kumar, 2015; Zhu et al., 2007).

In all of the studies, a peak around 10–30 nm was observed. Two studies reported unimodal distributions, three showed a second peak at 50–70 nm, and one exhibited a trimodal size distribution with an additional peak at 5 nm. As outlined above in the descriptions of traffic environments, these peaks can be attributed to primary emissions from gasoline engines (10–30 nm), nucleated particles from SVOCs emitted by diesel engines (10–30 nm), and primary emissions from diesel engines (50–70 nm) (Damayanti et al., 2023; Hopke et al., 2022).

4. Discussion

This review has examined and compared the PNC and PNSD characteristics of various transport and commuting environments. A total of 40 studies of ambient air in transport hotspots were selected, 31 of which were conducted in road traffic environments and nine of which were performed in or around airports. However, no studies meeting the review criteria were identified for harbors and railway stations. Our review also included nine studies that investigated passengers' exposure in various commuting modes, such as pedestrian, bicycle, bus, trams, subway, and in-vehicle travel. No studies were considering motorcyclist and aircraft passenger exposure to UFP. Thus, the first outcome of this review underscores the inadequacy of available information regarding both general and individual exposure to UFP concentrations and size distributions, particularly in specific transport modes. This gap in knowledge is attributed, at least in part, to the absence of air quality standards specifically addressing UFP and the challenges associated with conducting exposure assessments (Morawska and Wierzbicka, 2019; Ohlwein et al., 2019). Currently, PNC and PNSD measurements rely heavily on fixed monitoring stations; however, this approach may fail to capture the significant variations in concentrations that occur as individuals move closer to or further away from pollution sources and/or between different microenvironments (EFCA, 2019).

In terms of ambient air measurements conducted in fixed locations close to pollution sources, the average PNCs were higher in the vicinity of road transport environments (25,977 ± 20,494 #/cm^3 for continuous 24-hour long-term studies and 118,252 ± 116,440 #/cm^3 for diurnal short-term studies) than at airports (11,455 ± 3,598 #/cm^3 taking hours with no aircraft activity into account, and 30,654 ± 6,581 #/cm^3 when considering only hours with aircraft activity). The concentrations of UFP in airports exhibit significant variability throughout the day, primarily due to the intermittent nature of aircraft activities, with periods of no activities lasting approximately 4–6 h, depending on the airport. Additionally, the dispersion of contaminants in atmosphere through turbulent diffusion and wind transfer contributes to this variability (Zaporozhets and Synylo, 2019). Airports, typically situated in open areas with high dispersive atmospheric conditions, differ strongly from the more confined environments of many urban streets. The impact of airport activity appears to be most significant at a local level or in specific downwind locations (Stacey, 2019, and references therein). Indeed, Mazaheri et al. (2009) and Hsu et al. (2013) observed PNCs up to 2,000,000 and 7,000,000 #/cm^3, respectively, in close proximity to aircraft emissions during takeoff. However, rapid plume dilution and a subsequent decrease of PNCs to urban background levels were noted at a distance of only 500 m from the emission source (Hsu et al., 2013).

Although these findings suggest a potential concern for occupational exposure, especially for airport workers and travelers, further exposure assessment studies within different airport areas are needed in order to draw conclusions and make recommendations.

PNSDs at road sites typically exhibit a bimodal pattern with peaks at 10–30 nm and 40–90 nm. The smaller size mode is frequently attributed to spark-ignition vehicles (Hopke et al., 2022; Johnson et al., 2005; Kittelson, 1998) or the nucleation of SVOCs released from diesel engines (Damayanti et al., 2023), whereas the larger size mode is associated with primary emissions from diesel engines (Damayanti et al., 2023; Hopke et al., 2022; Dua et al., 1999; Kittelson, 1998).

Within airports, PNSDs were predominantly unimodal, with a major mode below 20 nm, which is smaller compared to particles emitted from road traffic. Ungeheuer et al. (2022) highlighted the significance of jet oil nucleation as a mechanism to explain the frequent observation of high PNC in non-refractory nucleation-mode particles, particularly within the range of 10–18 nm, near airports. Buonannoet al. (2012) observed a shift in the PNSD mode from 25 nm near the airstrip to 80 nm at a distance of 500 m, whereas Keuken et al. (2015) reported a unimodal PNSD with a peak around 10–20 nm at a downwind location 7 km from the airport. These observations highlight that PNSDs undergo dynamic changes over distance and direction from the emission source, indicating the need for further investigations into plume evolution and the transformation speed of UFP emitted by aircraft-related activities. It is crucial to emphasize that the growth of nucleation mode particles is not solely dependent on emission patterns but is also strongly influenced by atmospheric conditions such as humidity, photochemistry, and the concentration of gaseous precursors in the receptor atmosphere.

Only six studies at road traffic sites and one study near an airport provided separate data to facilitate comparisons between warm and cold seasons. Among road traffic studies, PNCs were found to increase by 1.34–2.8 times during winter relative to summer. The observed increase in PNCs during the winter months may be attributed to several factors. Lower ambient temperatures and higher relative humidity during winter intensify the nucleation of traffic vapors (von Bismarck-Osten et al., 2013; Kumar et al., 2011). Additionally, in winter, atmospheric conditions tend to be more stable, and the height of the mixing layer is lower, leading to reduced atmospheric dilution. The presence of additional sources, such as domestic heating, may also contribute to PNCs throughout the winter season.

Masiol et al. (2017) observed slightly higher PNC within the airport during the cold season (22,000 vs. 19,000 #/cm^3), accompanied by a slight shift toward larger particles in the PNSD (first peak: 15–24 vs. 14 nm; second peak: 70–100 vs. 30–50 nm). However, due to the absence of comparative studies, it is challenging to draw conclusive remarks regarding the influence of climatic conditions on PNCs and PNSD within airports.

Cars were found to be the commuting mode resulting in the highest degree of passenger exposure (27,643 ± 17,240 #/cm^3), whereas buses and subways exhibited the lowest PNCs (14,055 ± 7,951 #/cm^3 and 13,896 ± 2,681 #/cm^3, respectively). A comparison between the PNCs of the commuting modes is reported in Fig. 2. Active commuting modes, such as walking and cycling, resulted in intermediate exposure to PNC (19,481 ± 11,705 #/cm^3 and 22,660 #/cm^3, respectively), with lower concentrations observed when avoiding busier roads. Furthermore, PNCs in the commuting modes, including buses and trams, were found to be lower when outside air ventilation was reduced (Moreno et al., 2020; Ragettli et al., 2013). Additionally, PNCs in the subway microenvironment were predominantly influenced by the infiltration of outdoor traffic-generated air pollutants (Mendes et al., 2018; Moreno et al., 2020). None of the selected studies on commuting exposure reported separate data for warm and cold seasons and it is therefore not possible to interpret seasonal variability.

The PNSD observed during commuting were analogous to those identified at road traffic locations, which are characterized by mostly bimodal size distributions with peaks around 10–30 nm and 40–70 nm. These modes can be attributed to emissions from gasoline and diesel vehicles, as discussed above. The relative abundance of both of these modes exhibited considerable variability depending on the day and the specific means of commuting. Factors such as the occurrence of photochemical NPF bursts, particle growth rates, coagulation of fine particles, and potential effects of air filtration and recirculation within cars, buses, trains, and trams can influence the PNSD associated with road traffic during commuting.

Consistent with source apportionment studies attributing 71–94 % of PNSD in urban environments to road traffic (Hopke et al., 2022; Rivas et al., 2020), the PNCs observed in all commuting modes considered in this review appear to be predominantly influenced by road traffic. In-vehicle studies testing various ventilation settings found that the lowest exposure (up to 75 %) occurred with closed windows and air recirculation, thereby minimizing external air infiltration (Leavey et al., 2017; Goel and Kumar, 2015; Zhu et al., 2007). Zhu et al. (2007) emphasized two important points: i) the age of the vehicle, and, consequently, of the filter, significantly influences the ability to reduce concentrations, dropping to 25 % for older vehicles; and ii) recirculation leads to an increase in CO_2 concentrations, considered a practical proxy

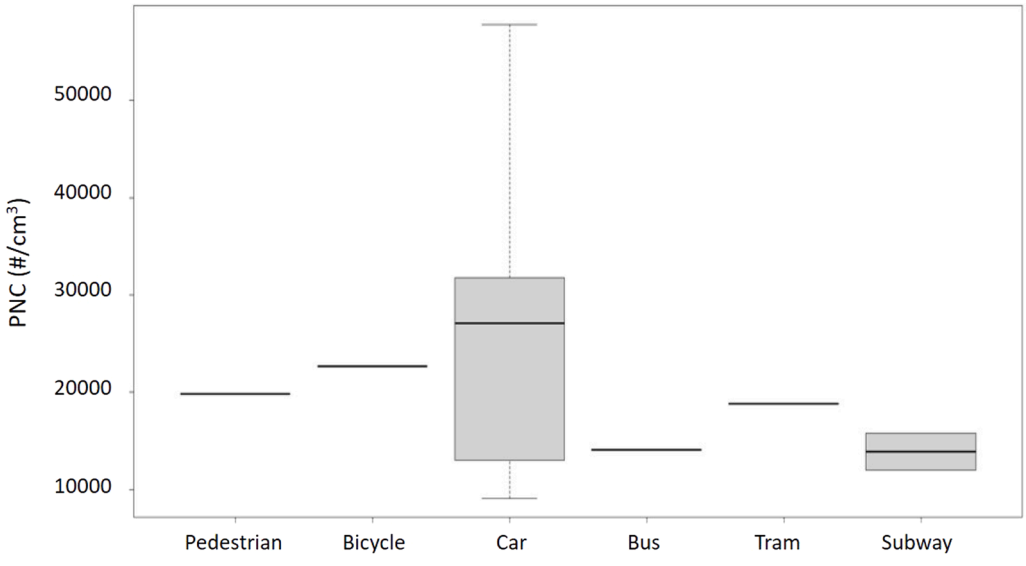

Fig. 2. PNCs associated with different commuting modes.

for ventilation and associated with the risk of infectious respiratory disease transmission (Rudnick and Milton, 2003, among others), from 420 ppm to 4,500 ppm in less than 10 min. Therefore, finding a balance between reducing the intake of outdoor air pollutants and ventilating with outdoor air to maintain acceptable CO_2 levels inside cars, buses, and trams, might require ventilating and filtering polluted outdoor air (Querol et al., 2022).

5. Conclusions

This review of 49 publications focused on particle number concentrations (PNC, considered equivalent to ultrafine particle concentrations, UFP) and particle number size distribution (PNSD) in different exposure hotspots related to transportation.

Unfortunately, the lack of information regarding personal exposure during various commuting and transport modes makes direct comparisons difficult. There is, therefore, a crucial need to carry out further PNC and PNSD measurements, especially during commuting and within airplanes and trains, where information is currently lacking.

This review also highlighted the absence of comprehensive studies of UFP and PNC-PNSD in harbor settings. Although shipping has been identified as the primary source of UFP in Europe, according to the "Dataset on ultrafine particles and non-exhaust sectoral emission distribution over Europe and pilot cities" (Kuenen et al., 2022), no eligible studies were found to meet the criteria of our review. Another challenge in making comparisons between transportation modes arises from the variability in particle size ranges measured by different studies. Indeed, even small differences in the lower detection limit can lead to significant variations in total PNC or nucleation mode (<25 nm) (Trechera et al., 2023, and references therein). It is strongly suggested to follow CEN recommendations available for both PNSD and PNC (CEN, 2020 and 2023, respectively). When using small portable devices, it is also recommended to intercompare (and correct if needed) the instrument with reference to these standard methods.

Acknowledging the lack of availability of data regarding harbors and railway stations, the highest PNCs inside cities in the reviewed articles were found at road traffic sites. Moreover, it was observed that PNC and PNSD in all analyzed commuting modes were related to proximity to road traffic and to the weather conditions (i.e., NPF events). In closed environments, such as cars, buses, trams, and subways, it has been observed that increasing the infiltration of external air to promote ventilation leads to an increase of PNC and a shift in PNSD toward smaller diameters. This shift results in bimodal size distributions with peaks at 10–30 nm and 40–70 nm; these characteristics are typical of gasoline and nucleation from diesel semivolatile organic compounds (SVOCs) and diesel primary emissions. Therefore, we emphasize the importance of measuring PNCs and PNSDs using instruments with lower detection limits of ≤10 nm. However, given the uncertainties related to ≤10 nm measurements (Kangasluoma et al., 2020), it is advisable to follow specific guidelines and protocols, including rigorous quality control measures, when selecting appropriate instruments for diverse applications.

At airports, very high concentrations were observed near runways, exhibiting a PNSD shift toward the nucleation mode, which was also detectable at great distances downwind from airports. However, since airport emissions occur in environments with high atmospheric dispersion, particles are rapidly diluted, resulting in lower average PNCs than road traffic sites. Our results also indicate that fast PNC-PNSD measurements are needed in these hotspots to detect aircraft UFP plumes.

Thus, as support for air quality policies, our review shows that:

- It is essential to measure UFP and PNC-PNSD starting from 10 nm (lower particle size detection) at all transport hotspots.
- High-concentration peaks observed within airports can pose an issue of occupational exposure and should be measured with rapid PNC-PNSD instruments.
- Studies are needed to evaluate similar parameters at harbors, in airplanes, at airport terminals, at railway stations, and on and around trains.
- There is a need for studies comparing PNCs and PNSD at the same location during warm and cold seasons in order to better interpret the influence of seasonal variability.
- A considerable distance should be maintained between road traffic and different commuting modes.
- Ventilation systems inside cars, buses, trams, and subways, necessary for reducing the risk of airborne transmission of pathogens, should include the filtration of outside air to diminish the entry of air pollutants such as UFP and NOx.
- Low-emission areas contribute to reducing PNC by prohibiting older vehicles that are not fitted with effective particle filtering systems.
- A general reduction in road traffic is necessary to minimize the daily exposure of citizens to UFP.

6. Limitations

Major limitations of this study include:

- This review is limited to 49 publications that provided appropriate data on UFP and PNSD in different exposure transportation-related hotspots. Further studies would strengthen the validity of the conclusions.
- The lack of personal exposure data from harbors, railway stations, motorbikes, and inside aircraft and trains limits the comparison of various commuting and transport modes.
- Variabilities in measurement settings (instrument type, lower particle detection size, period of measurement, and seasonality) also restrict comparisons. For example, according to Trechera et al. (2023), measurements from Helsinki show that PNC with particle sizes from 3 to 794 nm is reduced by 14 and 35 % when considering the range of 10 or 20–790 nm. However, differences <0.1 % are observed if the range extends from 3 to 594 nm.

Despite the effects of the above limitations, the fact that the highest exposure occurs close to road traffic remains valid, not only because of the high PNC, but also because of the frequency of exposure to UFP from this source.

CRediT authorship contribution statement

S. Ridolfo: Writing – review & editing, Writing – original draft, Visualization, Validation, Software, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **F. Amato:** Conceptualization, Funding acquisition, Investigation, Methodology, Project administration, Resources, Supervision, Validation, Writing – original draft, Writing – review & editing. **X. Querol:** Conceptualization, Funding acquisition, Investigation, Methodology, Project administration, Resources, Supervision, Validation, Writing – original draft, Writing – review & editing.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Fulvio Amato reports financial support was provided by European Commission. Xavier Querol reports financial support was provided by Spain Ministry of Science and Innovation and Government of Catalonia. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Since the paper is a review, we used only published information, which is already available

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Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envint.2024.108696.

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