Air pollution exposure in active *versus* passive travel modes across five continents: A Bayesian random-effects meta-analysis

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Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Highlights

- This systematic review assessed air pollutant exposures in users of 7 transport modes.
- Pairwise ratios of exposure were computed using a Bayesian meta-analysis approach.
- Users of motorized modes were the most highly exposed to gaseous pollutants.
- Public transport users were the most highly exposed to $PM_{2.5}$.
- Users of active mode were the most exposed to UFP.

Abstract

Epidemiological studies on health effects of air pollution usually estimate exposure at the residential address. However, ignoring daily mobility patterns may lead to biased exposure estimates, as documented in previous exposure studies. To improve the reliable integration of exposure related to mobility patterns into epidemiological studies, we conducted a systematic review of studies across all continents that measured air pollution concentrations in various modes of transport using portable sensors. To compare personal exposure across different transport modes, specifically active *versus* motorized modes, we estimated pairwise exposure ratios using a Bayesian random-effects meta-analysis. Overall, we included measurements of six air pollutants (black carbon (BC), carbon monoxide (CO), nitrogen dioxide (NO_2), particulate matter (PM_{10}, $PM_{2.5}$) and ultrafine particles (UFP)) for seven modes of transport (i.e., walking, cycling, bus, car, motorcycle, overground, underground) from 52 published studies. Compared to active modes, users of motorized modes were consistently the most exposed to gaseous pollutants (CO and $NO_{2)}$. Cycling and walking were the most exposed to UFP compared to other modes. Active vs passive mode contrasts were mostly inconsistent for other particle metrics. Compared to active modes, bus users were consistently more exposed to PM_{10} and $PM_{2.5}$, while car users, on average, were less exposed than pedestrians. Rail modes experienced both some lower exposures (compared to cyclists for PM_{10} and pedestrians for UFP) and higher exposures (compared to cyclist for $PM_{2.5}$ and BC). Ratios calculated for motorcycles should be considered carefully due to the small number of studies, mostly conducted in Asia. Computing exposure ratios overcomes the heterogeneity in pollutant levels that may exist between continents and countries. However, formulating ratios on a global scale remains challenging owing to the disparities in available data between countries.

Keywords : Active modes ; Particulate Matter ; Black Carbon ; Carbon Monoxide ; Nitrogen Dioxide ; Ultrafine particles ;

Introduction

The global burden of disease associated with air pollution is on par with major global health risk factors such as a poor diet and smoking (World Health Organization, 2021). In 2019, outdoor air pollution was responsible for 4.2 million premature deaths worldwide (World Health Organization, 2022). Studies have demonstrated a strong association between air pollution and major diseases, such as respiratory disease (Santos et al., 2021; Zhu, 2012), cardiovascular disease (Jerrett et al., 2013; Langrish et al., 2012), ocular disease (Lin et al., 2022), reproductive disorders (Boggia et al., 2009; Brauer et al., 2008) and cancer (Huang et al., 2021; Zhang et al., 2020). The International Agency for Research on Cancer (IARC) classified outdoor air pollution as a whole, and fine particles in particular ($PM_{2.5}$), as carcinogenic to humans, with sufficient evidence for lung cancer, and positive associations observed for bladder cancer (Hamra et al., 2015; IARC Working Group on the Evaluation of Carcinogenic Risks to Humans, 2012; Loomis et al., 2013; Turner et al., 2020).

Estimating chronic exposure to air pollution is a critical aspect of environmental epidemiology and health impact assessment. Due to a lack of information on individual activity and mobility patterns, epidemiological studies on health effects related to air pollution usually estimate exposures based solely on residential addresses (Reis et al., 2018; Steinle et al., 2013; Vanoli et al., 2024). This approach can lead to inaccurate estimates, especially when individuals spend time away from home (Poom et al., 2021; Setton et al., 2011). Traffic significantly contributes to air quality (Heydari et al., 2020), and while daily commute may represent overall a small proportion of daily time, it can result in a disproportionately high intake of air pollution (de Nazelle et al., 2013; Dons et al., 2012; Park, 2020; Van Ryswyk et al., 2021). Yu et al. (2020) reported that ignoring mobility in exposure estimation may lead to a 33% underestimation of relative risks (Yu et al., 2020). Ignoring daily mobility patterns may thus lead to effect estimates biased toward the null hypothesis in epidemiological studies (Park, 2020; Setton et al., 2011).

Importantly, personal exposures and inhaled doses during commuting may vary between transport modes, due to differences in micro-environmental characteristics and inhalation rates (Cepeda et al., 2017; Grana et al., 2017; Sommar et al., 2020; Zuurbier et al., 2009; de Nazelle et al., 2017). Walking has been reported as the least exposed mode compared to cycling, bus, or car for several pollutants based on several European studies, while bus users seem to be less exposed to black carbon compared to pedestrians (de Nazelle et al., 2017). Interest into active modes (i.e. cycling and walking) has substantially increased over the past decades because of their low to nil contribution to greenhouse gas and air pollution emissions, as well as personal physiological and health benefits associated with physical activity. (Brand et al., 2021; de Nazelle et al., 2011). Yet, the physical efforts and increased breathing frequency required by active transport may increase the inhaled air pollution dose in pedestrians and cyclists, estimated to potentially lead to up to 50% higher intake in a day compared to drivers (de Nazelle et al., 2012; Tainio et al., 2021). However, factors determining personal exposure to various air pollutants during travel associated with different modes of transport and their impacts on health at individual levels remain insufficiently understood (Poom et al., 2021; Ramel-Delobel et al., 2024). Moreover, a broad geographical coverage and thorough meta-analysis, of studies that used portable sensors to quantify personal exposure to air pollutants during travel in diverse locations is warranted (Poom et al., 2021).

To improve the integration of mobility patterns into epidemiological studies on health effects of air pollution exposure, this systematic review analyzed published studies worldwide that used portable sensors to quantify personal exposure to air pollutants (i.e., black carbon (BC), carbon monoxide (CO), particulate matter (PM_{10}, $PM_{2.5}$), ultrafine particles (UFP) and nitrogen dioxide (NO_2)) encountered by users of different urban transport modes (i.e., walking, cycling, bus, car, motorcycle, overground and underground), with a focus on active travel in comparison to other modes. We performed a Bayesian random effects meta-analysis approach to ensure the most robust estimate of exposure contrasts between modes (and their associated uncertainties). The purpose was to provide inputs for epidemiological studies or health impact assessments that wish to incorporate daily mobility in the characterization of individual exposure to major air pollutants.

1) Methods

1.1) Literature review

We conducted a systematic review on personal air pollution exposure for various modes of transport across continents. Exposure was defined by concentration levels to which an individual was exposed in a given microenvironment, based on portable sensor measurements, not by time-weighted concentration levels.

1.1.1) Search strategy

This review was carried out according to the PRISMA guidelines (2020) (Page et al., 2021). Studies published in peer-reviewed journals in English and French between the 1st of January 2000 and 5th of May 2022 were searched in PubMed, Scopus and Web of Science. The search was carried out between the 27th of February 2020 and the 6th of May 2022. The search strategy, detailed in Supplementary Materials A, used a combination of transport and air pollution terms. The electronic search was supplemented by hand-searching of references listed in the "cited by" sections common to the three research databases, and by the "similar articles" section of PubMed, "recommended articles" of Scopus, "you may also like" of Web of Science, as well as by a manual search in the reference lists of the identified publications.

1.1.2) Study selection

We selected eligible studies according to the following criteria:

1) Monitoring studies of air pollution concentrations in transportation microenvironments in Africa, Australasia, Asia, Europe, North America and South America;
2) Measurements with portable sensors for one or more of the main pollutants related to road traffic including BC, CO, NO_2, PM_{10}, $PM_{2.5}$ and UFP;
3) At least one active travel mode (walking or cycling) compared to one or more other travel modes (bus, car (i.e., private car, taxi), overground (i.e., tramway, train), underground (i.e., subway, metro, Mass Rapid Transit), motorcycle (i.e., motorcycle, scooter or auto rickshaw)); or at least one active travel mode compared to background concentrations. Studies indicating "public transportation" without further specifying the mode of public transport were excluded;
4) An experimental design including a comparison between modes on the same or close to the same routes, with concomitant or near-concomitant sampling for investigated modes;
5) Sufficient information to compute the parameter estimates (i.e., mean, standard deviation, sample size (i.e., number of measurements per mode for each study)) for the lognormal distributions (geometric means and geometric standard deviations). Studies with missing information for one of the following parameters were excluded: sample size, arithmetic mean and standard deviation (or geometric mean and geometric standard deviation) for each relevant mode of transport.

Studies estimating exposure exclusively by simulated data, or through biomarkers, as well as reviews, meta-analyses and systematic reviews, were excluded.

Two authors (MRD, TC) reviewed manually titles and abstracts of the studies identified in the three databases. Then, the full texts were screened independently by two authors (MRD, TC). Studies that did not meet the inclusion criteria were excluded and the reasons for exclusion were recorded for each study during the selection process. Any discrepancy was discussed and resolved by consensus between co-authors (MRD, TC, ADN).

To ensure the reproducibility of the analysis, only studies from which required data could be extracted directly from the main body and supplementary materials were included in the review (i.e., authors were not personally contacted to collect missing data).

1.1.3) Data extraction

Data were extracted according to each pollutant of interest, travel mode, continent, country and city, and when available, by ventilation parameters for vehicles (open/closed windows (OW and CW), air conditioning (AC) on/off, internal air recirculation (IR) on/off), season, month or time of day.

It should be noted that in this research each study could include two or multiple study campaigns (e.g., concentrations measured in a car with the windows closed and with the windows open, corresponding to 2 study campaigns; concentrations measured in April, August, and September, corresponding to 3 study campaigns, etc.).

For each mode of transport, the mean concentration (arithmetic mean, and geometric mean when available), standard deviation (from arithmetic mean, and geometric standard deviation when available) and sample size were extracted for each study campaigns. The supplementary materials were systematically consulted to complete the data extraction.

The studies were read several times to check the accuracy of the values collected. Uncertainties were resolved by the co-authors (MRD, TC, ADN).

1.2) Quantitative approach: Bayesian random effects meta-analysis

To identify which modes may expose commuters to higher or lower levels of a given air pollutant, we performed pairwise comparisons of concentration levels for a given pollutant between motorized modes of travel and active travel. We calculated concentration ratios for any pairs of interest, using active travel as the reference mode in the denominator (e.g., car vs walk, car vs cycle, bus vs walk, etc.). We obtained the pollutant concentrations reported in previous studies, harmonized the units, and calculated geometric means when this information was not reported.

Since between-study heterogeneity is highly plausible in this context (e.g., due to differences in exposure measurement methods), we adopted a Bayesian random effects meta-analysis approach, implemented in the statistical software WinBUGS (Lunn et al., 2000). A Bayesian random-effects meta-analysis is similar to its classical counterpart. However, instead of point estimates, we can obtain posterior densities for each parameter of interest under the Bayesian approach. This allows capturing uncertainties more fully. Note that our model-based approach using a meta-analysis and obtaining expected values of exposures to compute the ratios of interest, enabled estimates of uncertainties and produced more reliable statistical results compared to simply calculating crude ratios based on observed values. The latter approach corresponds to a fixed effect analysis that is incorrect in principal since it assumes all studies and study campaigns are identical in terms of their characteristics. While a fixed-effect meta-analysis does not allow for between-study (or study campaign) differences, a random-effects meta-analysis, as specified in equation (1) below, accommodates such differences effectively.

Assuming that the observed outcome of interest y_i (the geometric mean of exposure for a given pollutant in a specific travel mode reported by study campaign *i*) followed a normal density with varying means (expected values) and variances θ_i and σ_i^2, respectively, we carried out a meta-analysis as specified in equation (1). The means θ_i follow a normal distribution with the second stage variance τ^2 and the mean μ.

1.2.1) Prior specification

With small datasets in hierarchical meta-analyses, it is common for the between-study variance to have a large effect on the final estimates. This is because the variance cannot be well estimated with small datasets, so under the Bayesian framework the influence of the prior may remain. Therefore, it is important to do a robustness check on the prior for this variance. A non-informative flat prior is often assumed for the mean μ. To reduce sensitivity to prior choice when a relatively small number of studies were available, we considered a Pareto density based on the mean of σ_i^2 as suggested by previous research (Congdon, 2014; DuMouchel, 1996). Also, for selected scenarios with limited observations (underground-to-cycle ratio for BC (6 observations) and car-to-walk ratio for PM_{10} (11 observations)), we conducted sensitivity analyses using a weakly informative prior, Gamma (0.001, 0.001), on the variance. Let *n* denote the number of observations. To specify σ_i^2 the observed (actual) standard deviations from each study campaign (here, denoted by s_i) were considered as specified in the following equation (1):

$$
\begin{gathered}
y_i \sim Normal\ (\theta_i, \sigma_i^2) \\
\theta_i \sim Normal\ (\mu, \tau^2) \\
\sigma_i^2 = s_i * s_i \\
\tau = T - s_0 \\
\frac{1}{s_0^2} = \frac{1}{n}\sum_{i=1}^{n}\frac{1}{\sigma_i^2} \\
T \sim pareto(1, s_0)
\end{gathered}
\tag{1}
$$

Note that in (1) the observed (known) quantities were y_i and s_i, which were reported by each study campaign; the remaining parameters were to be estimated as part of the analysis.

1.2.2) Ratio estimation

To compare between any two modes of travel (e.g., driving and cycling), we estimated the ratio of expected exposure (θ_i) for the travel modes of interest (e.g., expected exposure while driving divided by the expected exposure while cycling). Within our Bayesian framework, the mean ratios for each pair of transport modes can be estimated for each study campaign and at each iteration of the Markov chain Monte Carlo (MCMC) simulations. Then, the overall average value of a given ratio was obtained with its corresponding 95% credible interval, representing the uncertainty around a ratio estimate.

Bayesian credible intervals have a straightforward interpretation in contrast to their classical counterparts, confidence intervals: a 95% credible interval indicates that an estimated ratio is in that interval with 95% probability. While we provided these intervals for all scenarios considered in our study to provide a full picture of the range of the ratios of interest, one may draw conclusive inferences only if an interval is either entirely on the right-hand side or on the left-hand side of one. In other words, if an interval includes the value one, it is unclear which mode has higher exposure. Ratios with credible intervals containing 1 are reported as statistically non-significant ratios.

Of note, the meta-analyses could not be performed at the continental level due to an insufficient amount of data, but for descriptive purposes median ratios were estimated for a pair of modes for each continent (e.g., car-to-walk ratio in Europe).

2) Results

2.1) Studies selected

Figure 1 shows the search and selection process for the present review. From a total of 356 citations identified from the literature, 236 were excluded after screening of titles (n=159) and abstracts (n=77). Examination of a total of 110 full-text studies led to the further exclusion of 68 studies: two reported only data from simulations; 19 did not allow for comparison of data between different modes or at least between an active mode and background concentration data; nine did not involve any active modes; and 38 did not provide sufficient information (mean, standard deviation and sample size) for meta-analysis.

Overall, 52 studies met the eligibility criteria and were included in this review. Supplementary Materials Table S1 provides detailed information on the 52 studies included in this review. More than two-thirds of the studies were carried out in Europe (n=19) and Asia (n=18), followed by North America (n=9), Africa (n=3), South America (n=2) and Australasia (n=1) (See Figure S1 in Supplementary Materials). Publication years ranged from 2001 to 2022, with a significantly larger number of studies (n=43) in the second decade of the study period (2012-2022).

Most of the studies assessed exposure to particles, including a majority of $PM_{2.5}$ measurements (n=49), followed by PM_{10} (n=16) UFP (n=16), and BC (n=14). Gaseous measurements included CO (n= 10) and NO_2 (n=3). Almost half of the studies considered only one pollutant (n=24), most of which were $PM_{2.5}$ (n=17); one third measured two pollutants (n=19); and a few, considered three (n=4) or four (n=5) pollutants, all including at least $PM_{2.5}$.

Regarding mode distribution, one-third compared cycling to motorized modes (n=18), one-third compared walking to motorized modes (n=16), and one-third included both cycling and walking (n=18). Cars (n=43) and buses (n=42) were the motorized modes most often compared to active modes. Less than half of the studies (n=22) included underground modes. Motorcycle and overground modes were less frequently included (n=8 and n=7, respectively). Finally, background measurements were available for one-fifth of the studies (n=12).

2.2) Range of pollutant concentrations

Mean concentrations in travel modes ranged for BC between 1.3 µg/m^3 (overground) to 14.1 µg/m^3 (bus), for CO between 0.4 ppm (underground) and 4.7 ppm (car), for NO_2 between 30 µg/m^3 (background) and 96.3 µg/m3 (car), for PM_{10} between 28.8 µg/m^3 (background) and 173.2 µg/m^3 (walk), for $PM_{2.5}$ between 31.7 µg/m^3 (overground) and 100 µg/m^3 (motorcycle), and for UFP between 17 473 part/cm^3 (overground) and 33 822 part/cm^3 (bus) (Table 1).

Across modes, observed ranges were highest for BC in buses (74 µg/ m^3), for CO in cars (32 ppm), for NO_2 in bicycles (85 µg/ m^3), for PM_{10} in motorcycles (520 µg/ m^3), for $PM_{2.5}$ in bicycles (331 µg/ m^3), and for UFP in cars (114 480 part/cm^3).

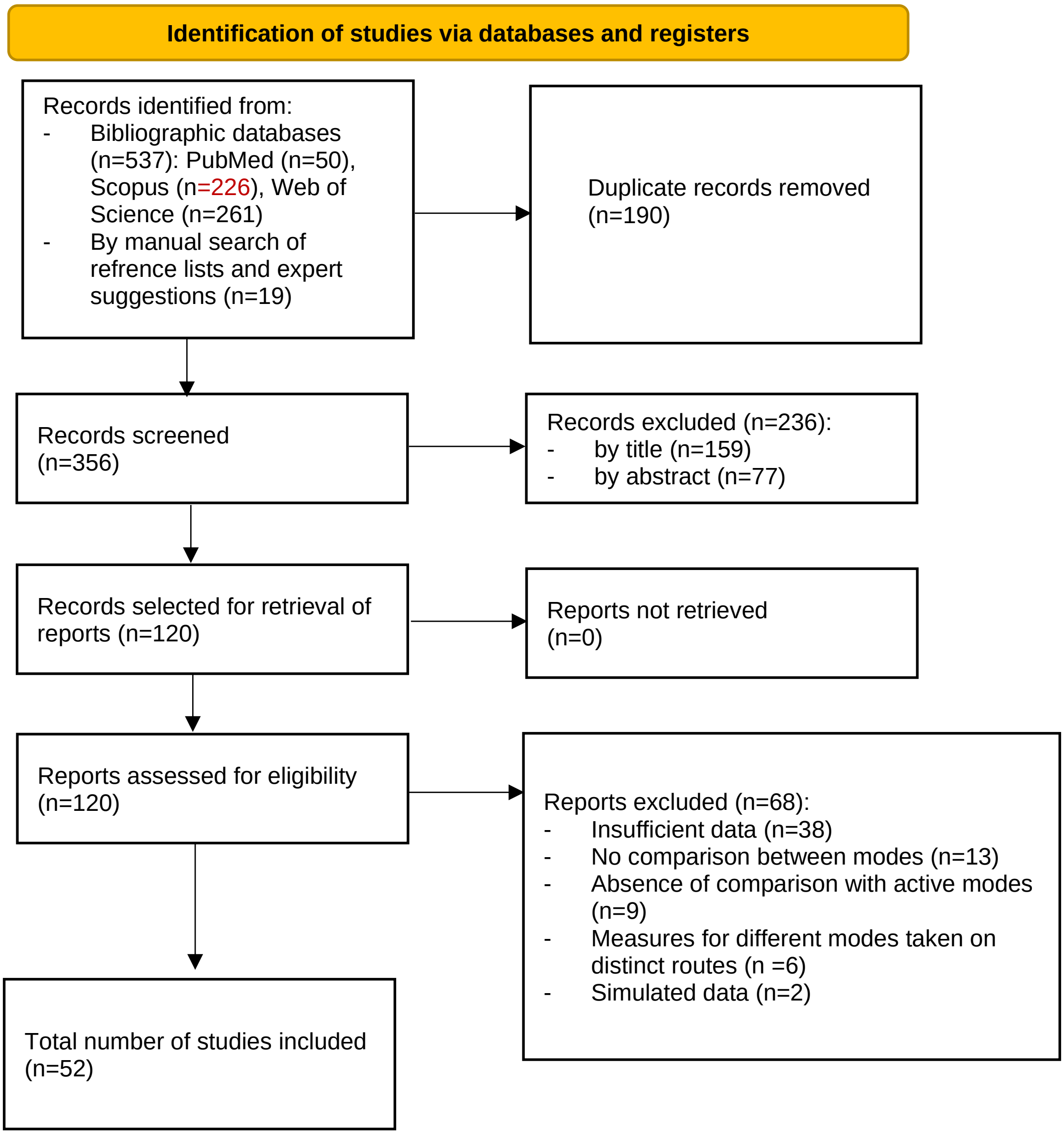

Figure 1: PRISMA flow diagram for screening and selection of studies

Table 1: Pollutant concentrations obtained from raw measurements (geometric mean) by mode of transport across studies

	BC (µg/m3)		CO (ppm)		NO_2 (µg/m3)		PM_{10} (µg/m3)		$PM_{2.5}$ (µg/m3)		UFP (part/cm3)	
	Sample size (nb of studies)	Mean (Min-Max)	Sample size (nb of studies)	Mean (Min-Max)	Sample size (nb of studies)	Mean (Min-Max)	Sample size (nb of studies)	Mean (Min-Max)	Sample size (nb of studies)	Mean (Min-Max)	Sample size (nb of studies)	Mean (Min-Max)
Background	8 (4)	1.8 (0.6-3.5)	4 (4)	0.6 (0.3-1.3)	3 (1)	30 (29.7-30.3)	3 (1)	28.8 (24.2-34.8)	16 (9)	77.9 (9.4-241.2)	14 (5)	17610 (8154-82819)
Bus	20 (12)	14.1 (0.8-74.7)	35 (11)	3.1 (0.1-23.2)	4 (1)	55.9 (35.9-62.8)	32 (13)	133.9 (18.6-384.6)	93 (36)	74.4 (2.1-298.8)	13 (9)	33822 (7451-99266)
Car	20 (10)	7.8 (0.1-41)	21 (8)	4.7 (0.2-31.8)	7 (2)	96.3 (81.8-108)	37 (15)	77.9 (5.8-425.7)	96 (36)	40.9 (2.2-155.5)	36 (13)	27162 (3120-117600)
Cycle	20 (12)	7.7 (0.7-41.7)	8 (5)	0.7 (0-1.8)	4 (4)	55.6 (27.6-112.8)	29 (13)	60.9 (6.2-126.4)	72 (33)	53.1 (4.2-334.9)	35 (13)	26248 (8398-88055)
Motorcycle			8 (3)	3.3 (0.2-16.3)			9 (4)	166.1 (26.9-547)	17 (6)	100 (21.1-228.3)		
Overground	2 (1)	1.3 (0.2-2.5)	4 (1)	0.8 (0.6-1)			1 (1)	37.2 (37.2-37.2)	9 (5)	31.7 (5.3-50.5)	4 (3)	17473 (4828-27333)
Underground	8 (6)	6.6 (2.4-10.6)	11 (4)	0.4 (0-1.7)			14 (6)	62.6 (34.3-125.6)	42 (21)	56.7 (19-238.7)	6 (5)	23168 (12425-39299)
Walk	7 (5)	13.3 (4.3-30.7)	15 (9)	2.1 (0.5-8.7)	6 (1)	35.1 (20.4-45)	12 (8)	173.2 (28.5-473.9)	51 (28)	60.9 (4.2-241)	10 (7)	30143 (11600-64861)
Total	85 (13)	8.9 (0.1-74.7)	106 (11)	2.7 (0-31.8)	24 (3)	59.2 (20.4-112.8)	137 (17)	98.6 (5.8-547)	396 (43)	59.1 (2.1-334.9)	118 (16)	26212 (3120-117600)

Mean concentration are calculated from the geometric mean. The sample size refers to the number of study campaigns used to calculate the mean concentration. BC = black carbon. CO = carbon monoxide. NO_2 = nitrogen dioxide. PM = particulate matter. UFP = ultrafine particles. Motorcycle = motorcycle, scooter or auto rickshaw. Overground = tramway, train. Underground = subway, metro, mass rapid transit.

2.3) Meta-analysis results

Estimated ratios to walk and cycle for various transport microenvironments are reported in the subsequent sections (Figure 2 and 3, Table 2). Note that due to a lack of data, ratios between the walk or cycle modes compared to the background could not be computed.

We used two chains for our MCMC simulations, each comprising 15,000 iterations. We discarded the first 5,000 iterations to meet convergence requirements according to the Gelman-Rubin convergence statistic (Gelman and Rubin, 1992). Therefore, the posterior inferences were based on 20,000 samples. Also, we carried out a sensitivity analysis to verify the robustness of our results in respect to prior assumptions. We did not observe any important differences (See the sensitivity analysis in Supplementary Materials Table S2). Note that the presence of occasionally wide credible intervals (or higher uncertainties) for certain ratios is due to the limited availability of studies for those specific ratios. As extra uncertainty exists in such scenarios, caution must be taken in drawing conclusions.

2.3.1) Ratios to walk

In general, pedestrians tended to be underexposed compared to users of other modes slightly more than they were overexposed (Figure 2). They emerged as being credibly underexposed compared to buses for PM_{10} (1.35, CI [1.31-1.41]), $PM_{2.5}$ (1.2, CI [1.15-1.28]), NO_2 (1.7, CI [1.66-1.75]) and CO (2.18, CI [1.35-12.45]); to cars for NO_2 (2.87, CI [2.82-2.93]); to bicycles for UFP (1.136, CI [1.136-1.136]) and $PM_{2.5}$ (1.34, CI [1.26-1.57]); and to motorcycles for PM_{10} (1.65, CI [1.63-1.66]) and CO (3.71, CI [1.82-26.29]). In reverse, pedestrians were credibly overexposed compared to buses for UFP (0.9141, CI [0.9139-0.9142]); to cars for UFP (0.9015, CI [0.9014-0.9016]), $PM_{2.5}$ (0.86, CI [0.79-0.99]) and PM_{10} (0.7, CI [0.68-0.73]); to bicycles for PM_{10} (0.62, CI [0.57-0.69]); to motorcycles for $PM_{2.5}$ (0.96, CI [0.94-0.99]) and to the underground for UFP (0.8097, CI [0.8096-0.8098]). All other combinations either generated credible intervals that included 1 and hence were not considered statistically significant, or could not be computed due to insufficient amount of data (Tables 2A and 3A).

Some ratios generated large credible intervals (e.g., car to walk ratio for CO: 1.97, CI [0.85-11.01]) and others more precise estimates (e.g., bus to walk for UFP: 0.9141, CI [0.9139-0.9142]; car to walk ratio for $PM_{2.5}$: 0.86, CI [0.79-0.99]), due to a limited number of samples (e.g., 4 samples for bus to walk for UFP) or low variability between studies (e.g., 79 samples for car to walk for $PM_{2.5}$).

In comparison with pedestrians and considering each pollutant separately (for statistically significant ratios), we observed varied exposure levels among different modes of transport. For $PM_{2.5}$ and PM_{10} (Figure 2), the exposure levels were equally distributed between under-exposed and over-exposed mode relative to pedestrians. Cycling showed higher exposures for UFP, while buses, cars and underground modes exhibited lower exposures relative to walking. Conversely, CO and NO_2 exposures were consistently higher across all modes (with buses and motorcycles for CO and buses and cars for NO_2). The greatest contrast for more exposed modes relative to pedestrians was for cyclists for $PM_{2.5}$ and UFP (ratios vs walking:1.34, CI [1.26-1.57] and 1.136, CI [1.136-1.136], respectively); motorcyclists for CO (3.7, CI [1.82-26.29]) and PM_{10} (1.65, CI [1.63-1.66]); and car drivers for NO_2 (2.87, CI [2.82-2.93]). (Tables 3 and 4). The greatest contrasts for modes with lower exposures than pedestrians were found for car drivers for $PM_{2.5}$ (0.86, CI [0.79-0.99]) and PM_{10} (0.7, CI [0.68-0.73]); and underground users for UFP (0.8097, CI [0.8096-0.8098]).

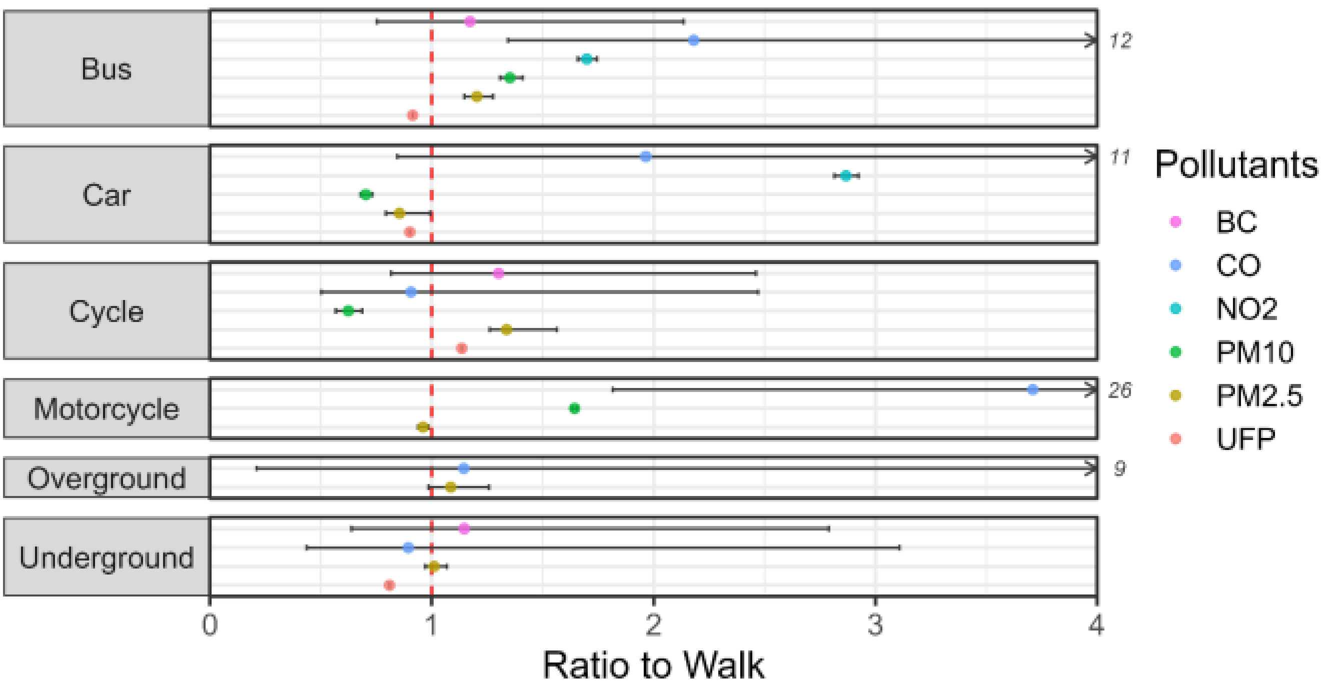

Figure 2: Ratios generated by Bayesian random-effects meta-analysis of pairwise comparisons using walking as the comparator

Forest plots represent the median. The limits of the error bars represent the credible intervals. Motorcycle = motorcycle, scooter or auto rickshaw. Overground = tramway, train. Underground = subway, metro, mass rapid transit. Ratios not shown in the figure were not computed because of insufficient data (see Table 2).

2.3.2) Ratios to Cycle

In general cyclists were shown to be more commonly less exposed compared to other mode users (Figure 3). Cyclists, in comparison to other modes (and retaining only statistically significant ratios), were shown to be underexposed compared to bus users for $PM_{2.5}$ (1.24, CI [1.19-1.47]), PM_{10} (2.3, CI [1.92-4.48]), and CO (5.03, CI [4.45-11.74]); to car drivers for CO (8.78, CI [7.82-21.18]) and BC (1.6, CI [1.12-6.29]); to motorcyclists for CO (4.23, CI [3.94-5.47]); to overground users for $PM_{2.5}$ (1.85, CI [1.59-2.21]) and for underground users for $PM_{2.5}$ (1.98, CI [1.88-2.11]), CO (2.37, CI [2.18-2.6]) and BC (2.01, CI [1.47-10.33]). Cyclists were overexposed compared to bus users and car drivers for UFP (0.8303, CI [0.8302-0.8303] and 0.9245, CI [0.9244-0.9245], respectively); motorcyclists for PM_{10} (0.69, CI [0.65-0.74]) and $PM_{2.5}$ (0.83, CI [0.8-0.86]) and underground users for PM_{10} (0.89, CI [0.87-0.91]).

Largest credible intervals were found (Tables 2B and 3B) (e.g., bus to cycle ratio for CO: 5.03, CI [4.45-11.74]) as well as others more precise estimates (e.g., motorcycle to cycle ratio for PM_{10}: 0.69, CI [0.65-0.74]; underground to cycle ratio for PM_{10}: 0.89, CI [0.87-0.91]) due to reduced number of samples (e.g., 5 samples for motorcycle to cycle for PM_{10}) or low variability between studies (e.g., 13 samples for underground to cycle for PM_{10}).

Considering specific pollutants (and only statistically significant ratios), cyclists were more commonly overexposed than underexposed for $PM_{2.5}$ (only motorcyclists had lower exposures) whereas the opposite trend was observed for PM_{10} (only bus users had higher exposures). For UFP both statistically significant ratios (bus and car) showed higher exposures for cyclists, and in reverse for BC both statistically significant ratios (car and underground) showed lower exposures for cyclists. All four exposure contrasts showed lower cyclist exposures for CO. The most extreme contrasts were found for CO, with median ratios ranging from 2.4 (underground vs cycle) to 8.8 (car vs cycle).

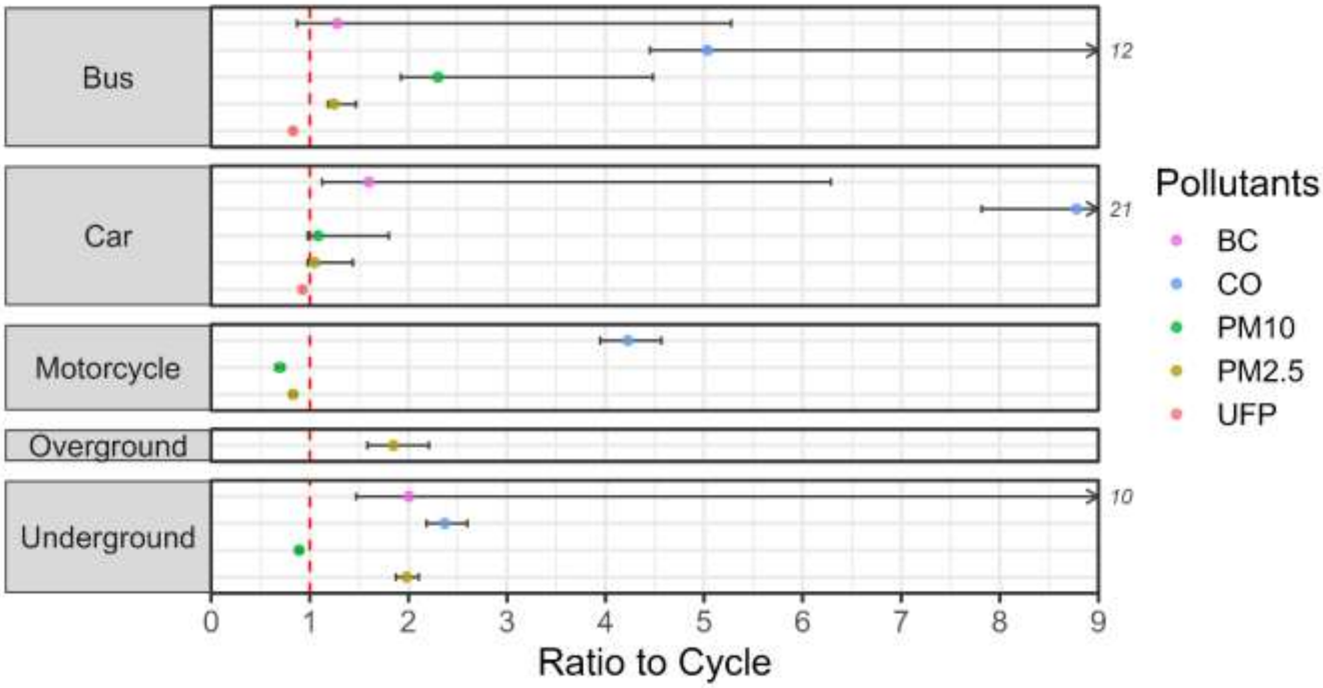

Figure 3: Ratios generated by Bayesian random-effects meta-analysis of pairwise comparisons using cycling as the comparator

Forest plots represent the median. The limits of the error bars represent the credible intervals. Motorcycle = motorcycle, scooter or auto rickshaw. Overground = tramway, train. Underground = subway, metro, mass rapid transit. Ratios not shown in the figure were not computed because of insufficient data (see Table 2).

Table 2: Ratios generated by Bayesian random-effects meta-analysis for particular matter, with walking (Table 2A) and cycling (Table 2B) as reference (denominator)

Modes	**$PM_{2.5}$**		**PM_{10}**	
A) To Walk	Nb of obs.	Median (95% credible interval)	Nb of obs.	Median (95% credible interval)
Bus	91	**1.2 (1.15-1.28)***	15	**1.35 (1.31-1.41)***
Car	79	0.86 (0.79-0.99)*	11	0.70 (0.68-0.73)*
Cycle	32	**1.34 (1.26-1.57)***	5	0.62 (0.57-0.69)*
Motorcycle	12	0.96 (0.94-0.99)*	5	**1.65 (1.63-1.66)***
Overground	7	**1.09 (0.99-1.26)**	/	/
Underground	26	**1.01 (0.97-1.07)**	/	/
B) To Cycle				
Bus	62	**1.24 (1.19-1.47)***	21	**2.30 (1.92-4.48)***
Car	80	**1.05 (0.98-1.44)**	27	**1.09 (0.98-1.80)**
Motorcycle	12	0.83 (0.80-0.86)*	5	0.69 (0.65-0.74)*
Overground	8	**1.85 (1.59-2.21)***	/	/
Underground	26	**1.98 (1.88-2.11)***	13	0.89 (0.87-0.91)*

Modes	**UFP**		**BC**	
A) To Walk	Nb of obs.	Median (95% credible interval)	Nb of obs.	Median (95% credible interval)
Bus	4	0.91 (0.91-0.91)*	4	**1.17 (0.75-2.13)**
Car	11	0.90 (0.90-0.90)*	2	/
Cycle	6	**1.13 (1.13-1.13)***	3	**1.3 (0.82-2.46)**
Motorcycle	/	/	/	/
Overground	2	/	/	/
Underground	6	0.81 (0.81-0.81)*	3	**1.15 (0,64-2,79)**
B) To Cycle				
Bus	8	0.83 (0.83-0.83)*	15	**1.28 (0.87-5.28)**
Car	27	0.92 (0.92-0.92)*	18	**1.6 (1.12-6.29)***
Motorcycle	/	/	/	/
Overground	1	/	/	/
Underground	2	/	6	**2.01 (1.47-10.33)***

Number of observations refers to number of crude ratios per study. Ratios greater than 1, indicating an overexposure of the mode to cycling or walking, are shown in bold. * indicates estimations statistically significant different to 1. Motorcycle = motorcycle, scooter or auto rickshaw. Overground = tramway, train. Underground = subway, metro, mass rapid transit.

Table 3: Ratios generated by Bayesian random-effects meta-analysis for gaseous pollutants, with walking (Table 3A) and cycling (Table 3B) as reference (denominator)

Modes	**CO**		**NO_2**	
A) To Walk	Nb of obs.	Median (95% credible interval)	Nb of obs.	Median (95% credible interval)
Bus	24	**2.18 (1.35-12.45)***	24	**1.70 (1.66-1.75)***
Car	10	**1.97 (0.85-11.01)**	36	**2.87 (2.82-2.93)***
Cycle	3	0.91 (0.50-2.47)	/	/
Motorcycle	5	**3.71 (1.82-26.29)***	/	/
Overground	4	**1.15 (0.21-9.01)**	/	/
Underground	6	0.89 (0.44-3.11)	/	/
B) To Cycle				
Bus	8	**5.03 (4.45-11.74)***	/	/
Car	7	**8.78 (7.82-21.18)***	/	/
Motorcycle	4	**4.23 (3.94-4.57)***	/	/
Overground	/	/	/	/
Underground	5	**2.37 (2.18-2.60)***	/	/

Number of observations refers to number of crude ratios per study. Ratios greater than 1, indicating an overexposure of the mode to cycling or walking, are shown in bold. * indicates estimations statistically significant different to 1. Motorcycle = motorcycle, scooter or auto rickshaw. Overground = tramway, train. Underground = subway, metro, mass rapid transit.

3) Discussion

This meta-analysis of studies published worldwide is the most comprehensive review to estimate exposure ratios between seven modes of transport (bus, bicycle, car, motorcycle, overground, underground and walk) and six pollutants (BC, CO, NO_2, PM_{10}, $PM_{2.5}$ and UFP). The ratios estimated using a Bayesian random effects meta-analysis allowed to establish robust exposure contrasts on observed pollutant concentrations for active modes (walking and cycling) compared to motorized modes (cars, motorcycles, buses, overground and underground). In comparison to active modes, motorized modes were consistently the most highly exposed to gaseous pollutants (CO and NO_2). For particulate matter, results varied according to particle size. For UFP, active modes were consistently more highly exposed compared to motorized modes. For particulate matter (PM_{10}, $PM_{2.5}$), active modes were less exposed compared to bus users. Cyclists tended to be the least exposed to BC and $PM_{2.5}$ (except for motorcyclists), but the tendency was reversed for PM_{10}. Of note, car drivers demonstrated overall lower exposures to particular matters than walking, while there were no statistically significant contrasts with exposures in cyclists. Motorcyclists were less exposed to $PM_{2.5}$ compared to both active modes, as well as to PM_{10} compared to cyclists. Conversely, they exhibited higher exposure to PM_{10} compared to pedestrians. The Bayesian meta-analysis showed also diverging results for rail modes with lower exposures compared to cyclists (PM_{10}) and pedestrians (UFP), and higher exposures compared to cyclists ($PM_{2.5}$ and BC).

3.1) Comparison with previous studies

a. Active modes

In the present study, cycling and walking were the most exposed to UFP among all modes. Cycling and walking were also the most exposed to $PM_{2.5}$ after public transport. These results differ from a previous review limited to ten European studies (Cepeda et al., 2017; de Nazelle et al., 2017), where active modes were less exposed than motorized ones. Active mode exposure is directly dependent on external variables such as the presence and intensity of emission sources and urban background concentrations (Qiu et al., 2019), and concentrations can vary from one country to another, depending on local emission sources, emission control measures, and vehicle fleets (Odekanle et al., 2016; Sarkan et al., 2017). This hypothesis seems credible in the light of the data for each continent. When restricting the present analysis to the 19 European studies, we observed a protective median ratio for active modes (see Supplementary Materials A Tables S3 and S4). Note that spatial heterogeneity is discussed further in the following section (see 3.2).

With regard to high exposure to active modes of transport, these might be mainly due to their direct proximity to vehicle emissions (Abbass et al., 2021; Moreno et al., 2015; Yang et al., 2021). Since active modes are often used in urban environments, their exposure depends also on the street canyon configuration which limits the dispersion of air pollutant concentrations (Betancourt et al., 2017; Ozgen et al., 2016), in addition to the traffic conditions that can be highly congested. It has been shown that the influence of heavy road traffic was associated with high particulate concentrations, due to particulate emissions from exhaust, tire and brake wear, and the resuspension phenomenon caused by vehicle motion (Fussell et al., 2022; Pant and Harrison, 2013). The higher particulate concentration during cycling could be explained by the road position of cyclists, who are often closer to vehicle exhaust compared to pedestrians (Huang et al., 2012; Ramos et al., 2016). For gaseous pollutants, the lower exposure compared to motorized modes could be explained by the open nature of the microenvironment in these modes, which facilitates the dilution of pollutants in the air (Ramos et al., 2016).

b. Car

Our results for CO and NO_2, showing higher exposure by car compared to active modes, confirm results from two previous reviews, i.e. CO (de Nazelle et al., 2017), and CO and NO_2 (Cepeda et al., 2017). Higher concentrations of gaseous pollutants by car compared to active modes are affected by the self-pollution emitted by the vehicles, as well as by emissions from neighboring vehicles that penetrate the cabin through the ventilation system (Ramos et al., 2016; Wong et al., 2011; Żak et al., 2017). Some studies have also shown that travel speed could be a dominant factor in CO emissions (Flachsbart, 1999; Sabapathy et al., 2012).

However, our results for PM and UFP remain equivocal when compared to previous reviews (Cepeda et al., 2017; de Nazelle et al., 2017). Concentrations in cars are shown in this review to be on average significantly lower than those for active mode users, even though it has been demonstrated that heavy road traffic is associated with high particulate concentrations inside cars (i.e., due to particles emissions by fuel combustion, tire friction on the road and brake wear, as well as infiltration of particulates due to proximity to vehicle exhausts) (Ramos et al., 2016).

The divergence in trends of exposure compared to active modes with the previous review by de Nazelle et al. (2017), reporting the highest exposure to $PM_{2.5}$ and UFP for cars, might be partly related to the vehicle ventilation parameters in older studies. Closing windows, using air conditioning or activating the internal air recirculation mode, can significantly reduce PM levels (Abbass et al., 2021; Chuang et al., 2013; Saksena et al., 2008). Of the 10 studies considered in the review by de Nazelle et al., five did not control ventilation parameters (Int Panis et al., 2010; Kaur et al., 2005; McNabola et al., 2008; Ragettli et al., 2013; Zuurbier et al., 2010), two measured exposure with CW and AC off (Boogaard et al., 2009; Gulliver and Briggs, 2004), two measured exposure with OW (Adams et al., 2001; de Nazelle et al., 2012) and one did not include cars in the modes considered (Moreno et al., 2015). In comparison of the 42 studies in our review that included the car in the measurements, 19 reported ventilation parameters, including 13 with the CW, 10 with the OW (8 studies measured both CW and OW), and 12 specified whether the AC or IR mode was activated. Also, in-cabin PM concentrations can vary between types of vehicles, fuel type (Raparthi and Phuleria, 2022; Zulauf et al., 2019), emission standards (Campagnolo et al., 2019; Cunha-Lopes et al., 2023) and efficiency of $PM_{2.5}$ filters (Abi-Esber and El-Fadel, 2013; Vande Hey et al., 2018).

c. Motorcycle

Motorcyclists were significantly more exposed to CO than cyclists and pedestrians. However, the difference between motorcycles and active modes is less pronounced than those found between closed modes (i.e., cars and buses) and active modes, due to the greater dilution of pollutants in an open microenvironment (Ramos et al., 2016).

In reverse, motorcyclists appear to be less exposed to $PM_{2.5}$ than cyclists and pedestrians, although motorcyclists are close and directly exposed to vehicle engines that produce high concentrations of PM (Maji et al., 2021; Manojkumar et al., 2021) and motorcyclists and cyclists have similar exposure conditions, as they are generally located at the roadside and can position themselves ahead of other vehicles (e.g., at traffic lights, during moments of congestion) (Goel et al., 2015; Vincent Wang et al., 2021).

Also, the results for PM_{10} show opposite trends depending on whether the ratios are considered in relation to walking or cycling. It should be noted that only eight studies for all pollutants, and five studies in the case of PM_{10}, considered motorcycles or similar modes (i.e., scooter or auto-rickshaw). Moreover, none of the studies compared motorcycling to both cycling and walking. As results, the ratios for PM_{10} were computed based on a few studies from different locations (i.e., the ratio for walking results from two Asian studies in India and Vietnam (Sabapathy et al., 2012; Saksena et al., 2008), and the ratio for cycling is from a single study in Portugal (Ramos et al., 2016)). These results ratios for motorcycles for PM_{10}, and for the other pollutants in this review, should therefore be interpreted with caution.

d. Underground

The substantial exposure to $PM_{2.5}$ and BC in the underground compared to active modes is consistent with the results of previous studies (Martins et al., 2015; Passi et al., 2021b; Querol et al., 2012). The high levels of PM in the underground can be explained by various factors, such as particle emissions due to metal wear (i.e., friction between wheels and rails, wear on brake pads, pantographs, catenaries, metal vaporization due to sparking and metro line maintenance, etc.), particle resuspension caused by the movement of passengers and trains in the station, and particle infiltration from outside air (Chang et al., 2021; Passi et al., 2021a). Numerous studies reported higher concentrations on the platform than in the vehicle (Barmparesos et al., 2016; Cartenì Moreno et al., 2015; Van Ryswyk et al., 2017), however only few studies showed the opposite trend (Braniš, 2006; Park and Ha, 2008; Ramos et al., 2016). These differences could be explained by the ventilation system and the age of the metro systems. Several studies have shown that the use of ventilation systems significantly reduces $PM_{2.5}$ levels (Cheng et al., 2008; Maji et al., 2021; Xu et al., 2016; Yang et al., 2021). In addition, newer and more modern subway systems also contribute to lower particle concentrations (Maji et al., 2021; Yang et al., 2021), where ventilation systems are advanced and screen door systems help reduce airflow from the tunnel to the platform (Cha et al., 2019; Font et al., 2019; Ji et al., 2021).

In addition, the composition of $PM_{2.5}$ concentrations in the underground mode is mostly composed of metallic components (mainly iron (Fe)) (Aarnio et al., 2005; Chang et al., 2021; Martins et al., 2016; Passi et al., 2021a) and carbonaceous matter (including BC – also called elemental carbon (EC)), and organic carbon (OC)) in old metro stations (Minguillón et al., 2018). However, high iron concentrations increase artificially BC concentrations measured by the instruments (Correia et al., 2020; Querol et al., 2012, p. 20212), as iron has light absorption at wavelengths similar to BC (Chow et al., 2001; Ji et al., 2021; Karanasiou et al., 2014; Moreno et al., 2015).

Finally, although the levels of particulate matter were high, the results indicated a low concentration of UFP for the underground modes compared to active modes, consistent with the findings of a previous study (Boniardi et al., 2021). Some studies explained this by the prevalence of indirect sources, such as the infiltration of ambient air into the station (Mendes et al., 2018)– which still increases the UFP levels, albeit slightly compared to other modes – and the absence of direct emission sources, such as combustion processes (Correia et al., 2020)

e. Bus & Overground

Bus and overground are road surface transports and their overall concentrations are influenced by their own emissions (Ham et al., 2017; Sabin et al., 2005; Wong et al., 2011) as well as those of other vehicles (Ham et al., 2017; Strasser et al., 2018). Also, since part of the public transport trip involves walking to and from stations or bus stops (Apparicio et al., 2018; Moreno et al., 2015), and waiting for the transit vehicle (Qiu and Cao, 2020), public transport exposure takes into account part of pedestrian exposure.

For several pollutants in our study, buses were among the three most exposed modes compared to active modes (i.e., PM_{10}, CO, NO_2, $PM_{2.5}$). These results are consistent with those observed in previous reviews (Cepeda et al., 2017; de Nazelle et al., 2017). For gaseous pollutants, bus-to-active modes ratios were lower than those for other motorized modes compared to active modes, such as cars. This is deemed to be related to the dilution of CO and NO_2, due to the larger interior volume of buses (Chan and Liu, 2001; Huang et al., 2012), intake from the roof instead of from the front as in cars (Abi-Esber and El-Fadel, 2013, 2013; Limasset et al., 1993) and airflows from the outside entering the vehicle when the doors open at stops (Huang et al., 2012; Jiao and Frey, 2014; Żak et al., 2017).

Conversely, the ratios compared to active modes for particles (PM_{10} and $PM_{2.5}$) were found to be higher for bus and overground modes compared to those in cars (Chaney et al., 2017; Huang et al., 2012; Jiao and Frey, 2014). This could partly be explained by the opening of doors, leading to the entry of outside air, increasing particle concentrations, added to the inflow and outflow of travelers resuspending particles in the vehicle (Huang and Hsu, 2009; Kumar et al., 2018).

The number of studies that integrate overground modes was smaller than those integrating buses. In addition, these studies mostly aimed at measuring the inhaled dose of the pollutants and did not focus primarily on the factors influencing the pollutant concentrations via these modes. However, we found a higher ratio for overground-to-active modes than for bus-to-active modes, being explained in the literature by more frequent stops (Ragettli et al., 2013) and in some cases, because measurements in overground modes were made with windows open (Che et al., 2016; Ragettli et al., 2013; Ramos et al., 2016).

Finally, the significant exposure attributed to bus and overground can also be linked to other factors found in the literature, such as external air leakage into the vehicle interior (Sabin et al., 2005), idling behavior (Hammond et al., 2007; Huang et al., 2012; Richmond-Bryant et al., 2009), as well as newer vehicles that tend to reduce bus and overground concentrations, through improved air filtration (Asmi et al., 2009).

3.2) Strengths and limitations

A major strength of the present review is the large number of studies (n=52) included assessing concentrations of air pollutants for modes of transport in different contexts and countries. The use of a Bayesian meta-analysis allowed pairwise comparison of ratios across studies enabling to compare ratios from different settings and identify significant differences in exposure between modes of transport.

Data heterogeneity

Data for active modes, buses and cars are generally from a large number of various studies for particulate matter ($PM_{2.5}$ and PM_{10}), as for CO, BC and UFP for some of the ratios (mode to walk or mode to cycle). However, data for NO_2 are very limited, as are data for rail and motorcycles for most pollutants. In addition, disparate data for cycling and walking make it difficult to rank cycling versus walking in terms of exposure to most pollutants. Finally, the paucity of background data makes it impossible to assess the exposure gap between background stations and at least pedestrian exposure, which is assumed to be closest to ambient concentrations.

Variation in study locations

Due to the differences between urban morphology, meteorology, pollutant emissions and background concentrations, which can strongly vary within the same city, variations in exposure between regions of the world are even more uncertain. The formulation of ratios on a global scale remains difficult owing to the heterogeneity of available data between countries (i.e., missing data for Australasia, reduced number or missing data for Africa and the Americas for certain pollutants etc.). Future monitoring studies are therefore strongly encouraged on continents where data are still limited. Also, authors wishing to use the values developed in this paper should note that most of the results are mainly from European (Western Europe and Finland, Poland, Greece and the European side of Turkey) and Asian studies (South and East Asia).

In addition, differences in mobility practices, whether geographically differentiated or linked to culture, also contribute to the differences in exposure between mode users across continents and countries. Northern European countries, where the cycling culture is very strong and reinforced by public policies, have built a large number of dedicated cycle paths away from road traffic, which has greatly reduced the exposure of cyclists in these countries (Noussan et al., 2020). A review by Kumar et al. (2018) also reported that the structure of roadways in Europe differs from those in American and Asian cities in that the sidewalk is separated from the road by barriers (i.e., vegetation or low boundary walls), which can explain the greater difference in measured concentration between cars and pedestrians found in Europe compared to the other continents (see Supplementary Materials Tables S3 and S4).

Furthermore, the historical development of transportation networks, which varies between countries and continents, is linked to the variation in exposure during travel, due to the age of the infrastructure, the technologies and the materials used. This is particularly true for underground lines, where Asian systems may differ from those in Europe and America, as they are generally newer and use platform screen doors (PSDs) and in-train air purifiers, resulting in lower pollutant concentrations (Kim et al., 2014; Kwon et al., 2016). These characteristics may therefore explain the wide range of BC concentrations, resulting in large credible intervals for the underground-to-active modes ratios. Additionally, the characteristics of vehicle types (e.g., fuel, age, brand), as well as urban morphology in diverse locations, can widely influence local emissions, leading to ratios with wide credible intervals for CO in relation to surface motorized modes (i.e., bus, car and motorcycle) (Teixeira et al., 2021).

Monitoring instruments

Differences in concentrations between studies might occur due to the choice of measurement devices. Among the 52 studies included in the present study, 40 used light-scattering particle sensors (LSPs) and only 5 included gravimetric instruments. Compared to light scattering devices, gravimetric instruments present a higher stability of measurements (although they cannot capture real-time pollutant concentrations as they require to weigh manually the filter to determine particle concentrations) (de Nazelle et al., 2017). However, Motlagh et al. (2021) demonstrated that particulate low-cost sensors were able to capture quite precise variations in concentration such as differences in routing, location in the vehicle and passenger flows (Motlagh et al., 2021).

For gaseous pollutants, electrochemical gas sensors were used in the majority of the studies for CO measures (8 out of the 10 studies) and in one study of the three studies on NO_2. Although their sensitivity remains less affected by temperature and humidity than other sensing technologies (e.g., solid-state metal oxide sensors (MOS) (Concas et al., 2021)), variation of ambient conditions (i.e., interfering gases and variation of temperature and relative humidity (HR)) may lead to inaccurate results, especially in the case of NO_2 (Sun et al., 2017).

Application of ratios in exposure assessment studies

The purpose of establishing ratios is to apply obtained ratios to ambient concentrations simulated by models. This approach facilitates the consideration of the impact of various transport modes on concentrations, particularly "closed" modes, in exposure assessments. In general, studies estimating pedestrian exposure to air pollution rely on concentrations inferred directly by models.

However, a study has shown that pedestrian exposure can vary among individuals within the same street (e.g., size differences between adults and children, position on the sidewalk, etc.) (Buzzard et al., 2009). In addition, since background-to-walk ratios could not be determined in our study, we cannot state with certainty that ambient concentrations accurately reflect pedestrian exposure. Nevertheless, the use of ratios-to-walk may allow future epidemiological studies to consider the relative variation between modes of transport during daily mobility, in the absence of actual background concentration levels.

Additionally, Park and Kwan (2017) highlighted the need to improve the accuracy of mode effects on inhalation (Park and Kwan, 2017). However, this must be coupled with the correct estimation of the concentrations specific to each mode microenvironment, before applying values related to physical activity-induced breathing volumes, as suggested in a recent study (Borghi et al., 2020). Therefore, establishing values such as ratios contributed to the accuracy of mode exposure variation, facilitating the integration of daily mobility into epidemiological studies.

Implication in policies promoting active travels

Promoting active modes of transportation over motorized ones has been shown to have beneficial effects on health (Nieuwenhuijsen and Khreis, 2016). A systematic review demonstrated that despite higher exposure to air pollution, users of active transportation gained an average of one year of life expectancy compared to users of motorized transportation (Cepeda et al., 2017). Moreover, previous health impact modeling studies and epidemiological analyses have shown that physical activity accrued during active travel contribute to a reduction in overall mortality risks for air pollution levels encountered in the large majority of cities worldwide. (Andersen et al., 2017; Juneja Gandhi et al., 2022; Mueller et al., 2015; Tainio et al., 2016). Consequently, benefits from physical activity generally outweigh the risk of air pollution in active transports, although the evidence remains insufficient across low- and middle-income countries, sensitive subpopulations (such as children, the elderly, pregnant women, and individuals with pre-existing conditions) (Jiang et al., 2023; Tainio et al., 2021). Additionally, these benefits are greater when the physical activity is performed in a low-polluted areas (Andersen et al., 2017; de Hartog et al., 2010; Kubesch et al., 2015; Sinharay et al., 2018). Therefore, it is essential to include physical activity in health assessment studies to comprehensively quantify the health impact of active transportation modes and to further support measures aimed to reduce pollutant emissions.

As more individuals choose environmentally sustainable modes of transport, such as active modes, over traditional motorized transport options, there is a notable decrease in harmful emissions, particularly greenhouse gases, nitrogen oxides and particulate matter (Brand et al., 2021; Rodrigues et al., 2020). Although users of active modes contribute the least to road traffic emissions, they appeared to be the most exposed to particulate matter in our study. Our results therefore highlight the need to protect users of active modes from road traffic emissions to reduce the observed exposures (Shekarrizfard et al., 2020).

Urban planning and policy initiatives should prioritize the promotion and support of active transportation by developing safe and accessible walking and cycling paths, bicycle lanes, and public transportation networks (Dalton et al., 2013). Additionally, traffic management strategies can enhance support of active transportation by reducing emissions that users are exposed to, while also improving security and allocating more space for active mode users. A literature review comparing the effects of various traffic management strategies on emissions, air quality, exposure, and health found that restrictive road traffic measures, such as low emission zones or area road pricing, were the most effective in improving air quality compared to other measures (e.g., speed management, lane management)(Bigazzi and Rouleau, 2017).

However, careful implementation is essential to mitigate unintended health impacts of these traffic management strategies, such as the shift of traffic emissions from restricted areas to the outskirts (Nieuwenhuijsen and Khreis, 2016), and effects beyond pollution effects, such as socio-spatial segregation of individuals with reduced mobility (Paradowska, 2018). Consequently, measures aimed at promoting active modes of transportation and reducing road traffic emissions, by limiting motorized vehicles in urban areas must consider the socio-economic impact of such restrictions. This comprehensive approach is essential for improving the overall health of the population, beyond merely reducing exposure to air pollution.

Conclusions

This systematic review and meta-analysis of 52 studies published worldwide allowed to calculate exposure ratios between five motorized modes and two active modes and for six major air pollutants. Using Bayesian random effects meta-analysis, the ratios showed that motorized modes were the most exposed to gaseous pollutants, while active modes were the most exposed to UFP. For public transport, exposure was influenced by both vehicle characteristics and exposure during walking transit, resulting in higher exposure to $PM_{2.5}$ compared to active modes, particularly for underground mode.

While transport-related exposure patterns varied between continents, these ratios produced by Bayesian approach allowed to overcome the differences in background pollutant levels that may exist between continents and countries, and thus, provide valuable data for future use in epidemiological studies aiming to integrate daily mobility into air pollution exposure estimates. Further studies are needed in countries where data are still scarce, in order to further increase the representativeness of the ratios produced.

Additionally, the results of this study highlighted the need to protect users of active modes from road traffic emissions, by adapting urban planning and transport policies to promote and support active transportation options, including the development of safe and accessible walking and cycling paths, bike lanes, and public transportation networks.

CRediT authorship contribution statement

Marie Ramel-Delobel: Conceptualization, Methodology, Validation, Formal Analysis, Investigation, Data Curation, Visualization, Supervision, Writing – Original Draft.

Shahram Heydari: Methodology, Formal analysis, Data curation, Software, Writing- Original Draft.

Audrey de Nazelle: Conceptualization, Methodology, Resources, Validation, Supervision, Writing – Review & Editing.

Delphine Praud: Writing – Review & Editing.

Pietro Salizzoni: Supervision, Writing – Review & Editing.

Béatrice Fervers: Conceptualization, Methodology, Supervision, Funding Acquisition, Writing – Review & Editing.

Thomas Coudon: Conceptualization, Methodology, Validation, Investigation, Data Curation, Project Supervision, Visualization, Administration, Funding Acquisition, Writing – Review & Editing

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Data availability Statement

The data presented in this study are available on request from the corresponding author.

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