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Personal Exposure to PM_{2.5} in the Megacity of Mexico: A Multi-Mode Transport Study

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Abstract: Recurrent personal exposure to ambient PM_{2.5} is associated with adverse human health effects, in particular on the respiratory and cardiovascular systems. Here, we present an assessment of personal exposure and inhalation of PM_{2.5} for five modes of transport (walking, cycling, public bus (trolleybus and diesel bus), conventional car (CC) and hybrid-electric car (HEC)) and two routes of similar distance, along a major road in the Mexico City metropolitan area (MCMA). Arithmetic average exposure concentrations ranged from $16.5 \pm 6.5 \mu\text{g m}^{-3}$ for walking to $81.7 \pm 9.1 \mu\text{g m}^{-3}$ for cycling (henceforth shown as average ± 1 SD), with no significant differences with geometric averages. The maximum exposure concentration of $110.9 \mu\text{g m}^{-3}$ was observed for the conventional car. The highest exposure concentrations depended on route and the mode of transport, being observed for cycling and walking. The PM_{2.5} measurements showed large spatial heterogeneity in the exposure levels for walking and cycling, while public buses and private transport showed less spatial heterogeneity. The greatest peaks in PM_{2.5} coincided with 4-way intersections for all modes of transport, being positively influenced by traffic density. The mass of PM_{2.5} inhaled depended mostly on the mode of transport, and ranged between 1.0 ± 0.3 and $30.1 \pm 14.2 \mu\text{g km}^{-1}$ for the HEC and bicycle, respectively. Local area PM_{2.5} increments identified as ‘residuals’ after subtraction of data recorded at the closest fixed monitoring site from exposure concentrations along the studied road suggested that inhalation for bicycle and diesel buses is strongly influenced by vehicular emissions. Residuals estimated for the trolleybus, CC and HEC confirmed a lower inhalation than for the other modes of transport evaluated due to protection by the cabin.

Keywords: air quality; cyclists; inhalation; pedestrians; vehicular emissions

1. Introduction

Immediate proximity to direct emissions of PM_{2.5} from motor vehicles during daily commutes represents a significant threat to human health. Although personal exposure in transport micro-environments represents only between 1–5% of the time spent in a day [1,2], in large urban areas this time may increase

significantly. For example, within the Mexico City Metropolitan Area (MCMA), a typical single journey may take between 41 and 81 mins, while average journeys of 51 mins represent spending 16 h per week in transport micro-environments [3]. Therefore, longer commuting times may result in an increase in exposure to PM_{2.5} which can be significantly relevant for health effects [4]. For instance, Dons et al. [5] reported a difference of 2-fold in personal exposure to black carbon for commuters in Belgium compared with non-commuters. Within the Greater London Area, Smith et al. [2] reported that daily commutes contribute around 9% of total daily exposure to PM_{2.5}. However, this can be further exacerbated during traffic rush hours, when exposure concentrations of PM_{2.5} may increase by between 5–20% [6,7].

Besides proximity to emission sources and commuting time, personal exposure to PM_{2.5} may depend on the mode of transport, being classified into active (cycling and walking) and passive (car, bus, underground) [1,8]. For instance, Okokon et al. [9] addressed exposure to PM_{2.5} when commuting in cars with open and closed windows, and by bicycle and bus in three European cities, observing mostly higher exposure for passive transport modes. Similarly, higher exposure to PM_{2.5} in Central London for passive as compared to active commute was reported by Adams et al. [1]. In the MCMA, studies have addressed personal exposure to carbon monoxide with the highest and lowest levels observed for private and public transport, respectively [10], and to volatile organic compounds with exposure levels to vehicular related compounds being a factor of 2 greater than observed indoors [11]. Vallejo et al. [12] addressed personal exposure to PM_{2.5} during transportation within the MCMA by underground, public bus and private car, reporting median exposure levels of 106.2, 101.7 and 62.4 µg m⁻³, respectively; however, their study design did not allow for comparison between micro-environment concentrations. This highlights the importance of assessing personal exposure to PM_{2.5} during daily commutes using comparable information of modes of transport and routes.

In this work, we describe personal exposure to PM_{2.5} of commuters using five modes of transport along a typical main road of the MCMA. We estimate commuter inhalation by taking into account physical activity to provide a better approximation of the effect of the transport micro-environments upon mass of PM_{2.5} inhaled. We compare commuters' inhalation calculated from in-situ measurements of PM_{2.5} along the studied road with those from a fixed site to account for the effect of road proximity. Finally, we describe the traffic effect on the exposure to peaks in PM_{2.5} by showing spatial and temporal variability in recorded data.

2. Methodology

2.1. Study Site Location and Description

Personal exposure to local background and proximity to sources of PM_{2.5} as a function of the transport mode were evaluated along the Miguel Angel de Quevedo (MAQ) Road in South MCMA (Figure 1a). The MAQ Rd was selected due to the circulation of different modes of transport, proximity to a ground-based monitoring site, and confined lanes and a return route, which offered two routes for the same mode of transport. It is located in the Coyoacán municipality and is one of 10 main roads that cross from west to east in the southern region of the MCMA (Figure 1b). It runs through a densely populated residential area, with three lanes in each direction separated by a strip of trees and grass. The right extreme lanes are mostly designated for cyclists and public transport, consisting of diesel buses and trolleybuses. Measurements of personal exposure to ambient PM_{2.5} were made along 2 routes as shown in Figure 1: (i) From the MAQ roundabout to the Taxqueña (TAX) trolleybus stop (eastwards: MAQ-TAX) and (ii) from TAX to MAQ (westwards: TAX-MAQ). Table 1 lists the points of departure and arrival of each route and corresponding length. Figure 1a also shows the location of the ground-based Coyoacán (COY) monitoring site, which was used as reference of personal exposure in the local area. The COY site forms part of the Integral Atmospheric Monitoring System (SIMAT) of the Mexico City Government. PM_{2.5}, wind speed (WS) and wind direction (WD) have been monitored

continuously since 2003 at the COY site and the data sets were obtained from the SIMAT web site (Available online: <http://www.aire.cdmx.gob.mx>).

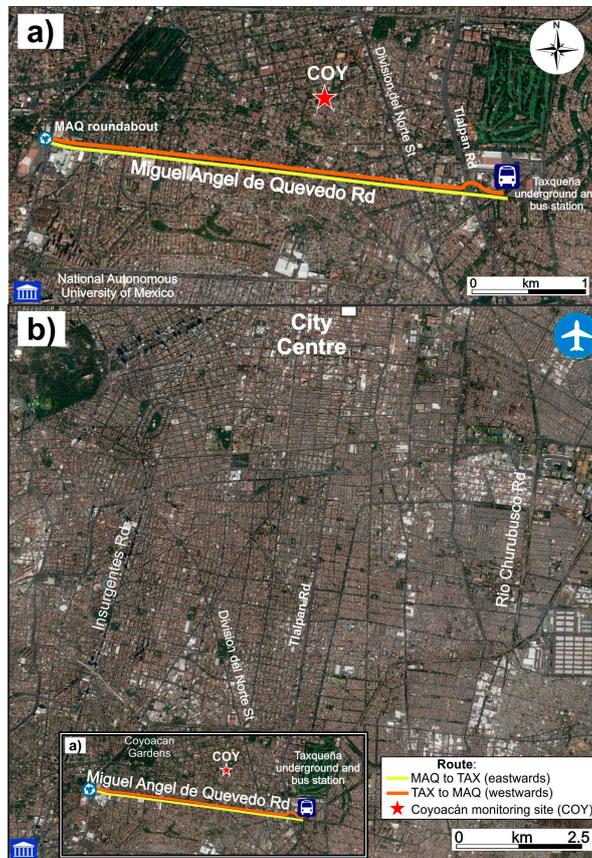


Figure 1. (a) The Miguel Angel de Quevedo (MAQ) Rd and the Coyoacán (COY) monitoring site close to MAQ Rd in the local context; (b) The MAQ Rd in relation to the whole Mexico City Metropolitan Area (MCMA).

Table 1. Transport modes and distances travelled. (MAQ: Miguel Angel de Quevedo; TAX: Taxqueña; CC: conventional car; HEC: hybrid-electric car).

Transport Mode	MAQ-TAX				TAX-MAQ			
	From:	To:	Distance (km)	Duration (h)	From:	To:	Distance (km)	Duration (h)
Walking	MAQ roundabout	Division del Norte Rd	3.5	1.07	Division del Norte Rd	MAQ roundabout	3.3	1.02
Cycling	MAQ roundabout	Division del Norte Rd	3.8	0.51	Division del Norte Rd	MAQ roundabout	3.5	0.49
Trolleybus	MAQ roundabout	TAX trolleybus station	4.2	0.3	-	-	-	-
Diesel bus	-	-	-	-	Bus station	MAQ roundabout	4.3	0.36
CC	MAQ roundabout	Division del Norte Rd ^{1,2}	4	0.35	Division del Norte Rd	MAQ roundabout ³	3.9	0.34
HEC	MAQ roundabout	Division del Norte Rd ^{1,2}	4.5	0.39	Division del Norte Rd	MAQ roundabout ³	4.5	0.37

¹ Trip with closed windows and operating air conditioning; ² Data discarded due to increases in relative humidity inside the vehicles (>85%), which affected instrumentation; ³ Trip with open windows.

2.2. Study Design

The measurement campaign was carried out during 10–14 November 2014, between 11 a.m. and 2 p.m. CDT, when traffic loading allows capture of features relevant for personal exposure and inhalation, i.e., vehicular accumulation in 4-way intersections, traffic lights and different traffic composition [13]. One journey along two routes (MAQ-TAX and TAX-MAQ) was performed for walking and cycling, and one for trolleybus (MAQ-TAX), diesel bus, conventional car (CC) and hybrid-electric car (HEC) (TAX-MAQ). Although all journeys were made along the MAQ Rd, MAQ-TAX and TAX-MAQ, the journeys were analysed separately as differences in variance coefficients for journeys duration, which were used here as proxy for traffic conditions, and were $\leq 10\%$ for all modes of transport apart from public bus (Supplementary Information S1.1, Table S1). This allows discarding the effect of traffic conditions on the personal exposure levels observed. Analysing the routes separately permits addressing the effect of traffic composition, time of day and meteorology. Walking exposure concentrations were measured by pedestrians moving at an average speed of 3 km h^{-1} . Cyclists made journeys on 18 gear, full suspension mountain bicycles at variable speed between 8 and 10 km h^{-1} depending on traffic conditions. For the public bus's eastward journey, volunteers travelled on a trolleybus (K route), whereas for the westward public bus journey, the volunteers travelled on a diesel-powered bus (1–29 route). Journeys by CC and HEC on the eastward route were made with closed windows and operating air conditioning, whereas westward journeys were made with open windows. Journeys between MAQ to TAX on the HEC and CC with closed windows and operating air-conditioning were discarded due to increases in relative humidity inside the vehicles ($>85\%$), which affected instrumentation. Table 1 describes the experimental setup.

2.3. Instrumentation

The exposure concentrations to $\text{PM}_{2.5}$ were measured using portable aerosol real-time photometric monitors (pDR-1500, Thermo Scientific, Franklin, MA, USA), operated at a frequency of 1 Hz, with stated precision for 30-days of $\pm 2\%$ ($\pm 0.005 \text{ mg m}^{-3}$) at $(2\sigma)^2$. During all journeys, 2 monitors were carried on the chest of volunteers and operated simultaneously. The monitors were operated at a flow rate of 2 L min^{-1} and calibrated with a commercial Agilent digital flow meter at a variance coefficient $<5\%$. Calibrations to zero were carried out inside a closed chamber filled by a zero-air Teledyne generator 701, operating at a pressure of 5 psi. Overall, the monitors were placed inside the closed chamber for 15 min to obtain a stable reading of zero. For all modes of transport, there was a very good agreement with $R^2 > 0.9$ between the two instruments, which suggests no significant differences in linearity between the two instruments over the exposure levels measured (Supplementary Information S1.2, Figure S1). Location of each volunteer along the MAQ Rd was recorded using a Hemisphere GPS receiver (60CSx, Garmin, Olathe, KS, USA).

2.4. Data Analyses

The personal exposure to $\text{PM}_{2.5}$ for each mode of transport was determined by averaging the data of both pDR-1500. Exposure concentrations were allocated spatially by merging $\text{PM}_{2.5}$ with GPS data. The data sets were analysed extensively using the R software [14], and tested for normality using the Shapiro-Wilk test for large samples ($n > 1000$), with descriptive statistics reported accordingly. Spatial variations in concentrations of $\text{PM}_{2.5}$ were depicted using the Google Earth software 2017 [15]. The determined $\text{PM}_{2.5}$ 1-s averages concentration maps exhibit a picture of main $\text{PM}_{2.5}$ high exposure areas along the MAQ Rd, where each point represents 1-s averaged data.

2.5. Residual Exposure Determination

We defined a 'local area residual' as the excess $\text{PM}_{2.5}$ in measurements along MAQ Rd compared to the contemporary COY levels, i.e., COY 1-h averages are subtracted from $\text{PM}_{2.5}$ exposure concentrations along MAQ Rd. The COY site is located upwind of MAQ Rd as shown in Figure 1.

2.6. PM_{2.5} Inhalation Calculation

We calculated the inhaled mass of PM_{2.5} for each mode of transport, considering the whole exposure windows experienced during the entire journeys. We defined an exposure window as the exposure concentrations experienced (recorded) over the duration of one period of inhalation-exhalation, here called “breath”. To allow a better interpretation, units for each calculated variable have been included. Thus, we first calculated duration by breath (b) as:

$$b [\text{min breath}^{-1}] = \frac{1}{RR} \left[\frac{1}{\text{breath min}^{-1}} \right] \quad (1)$$

where RR represents the respiratory rate (i.e., number of breaths per min). Therefore, the period for one breath (p) was calculated as

$$p [\text{min}] = 1 [\text{breath}] \times b [\text{min breath}^{-1}] \quad (2)$$

We assumed that the duration of each exposure window is equal to the period, and depends on the physical activity. Table 2 lists the duration of each exposure window, establishing lower and upper bounds by rounding p . Then, the number of exposure windows experienced during the whole journey (W) is given by

$$W = \frac{T}{p} \left[\frac{\text{min}}{\text{min}} \right] \quad (3)$$

where T represents the journey duration for each mode of transport. Then, the volume of air inhaled (V) in each exposure window was obtained by dividing the volume of air inhaled per minute (V_E) according to physical activities by the RR :

$$V [m^3 \text{breath}^{-1}] = \frac{V_E}{RR} \left[\frac{m^3 \text{min}^{-1}}{\text{breath min}^{-1}} \right] \quad (4)$$

Table 2. Respiratory parameters used in the current study to estimate inhalation of PM_{2.5} by mode of transport.

Transport Mode	Exposure Window Duration in s (p) *	Respiratory Rate in min ⁻¹ (RR)	V_E in L min ⁻¹
Diesel bus	[4,5]	13	11.4
Trolleybus	[4,5]	13	11.4
CC and HEC	[4,5]	15	14.3
Walking	[3,4]	18	18.3
Bicycle	[1,2]	34	42.1

* Lower and upper bounds were determined by rounding p . Note that for the reader convenience p is expressed in seconds.

The exposure concentration in the i exposure window ($i \in [1, W]$) was obtained by averaging the concentrations recorded (n) during one breath:

$$\bar{C}_i \left[\text{breath } \mu\text{g m}^{-3} \right] = \frac{C_{i,1} + \dots + C_{i,n}}{n} \left[\frac{\mu\text{g m}^{-3}}{\text{breath}^{-1}} \right] \quad (5)$$

Finally, the total mass inhaled of PM_{2.5} (I) was calculated as the integrated inhalation for all exposure windows experienced during the whole journey [16,17] (Supplementary Information S1.3, Table S2), as:

$$I [\mu\text{g}] = \sum_{i=1}^W (\bar{C}_i V) \left[\frac{\mu\text{g m}^{-3}}{\text{breath}^{-1}} m^3 \text{breath}^{-1} \right] \quad (6)$$

The RR and V_E for a cross-sectional study of healthy adults and physical activity within the MCMA were obtained by the National Institute of Respiratory Diseases of Mexico. Table 2 lists average RR and V_E were recalculated from data within the standard deviation (1σ) of the average calculated from the whole sample. Overall, Mexican men and women between 10 and 80 years of age, who were residents in the MCMA for more than 2 years were selected on the basis of pulmonary health and sedentary and low-moderate physical daily activity. High performance athletes, and patients with cardiopulmonary disease, obesity, physical limitations, and who smoked, were excluded from the cross-sectional study. Anthropometric evaluations, forced spirometry, maximum voluntary ventilation and an electrocardiogram in resting conditions were carried out for all volunteers. The analysis of the exhaled air was made using oronasal mask. A protocol of incremental exercise of 10 W min^{-1} symptom-limited was carried out using a cycle ergometer, monitoring cardiovascular, respiratory and metabolic responses. The cardiopulmonary tests were made with Jaeger Oxycon-Pro instruments.

3. Results

3.1. Time-Series of Personal Exposure to $PM_{2.5}$

$PM_{2.5}$ data were recorded for five transport modes from TAX to MAQ and for three modes from MAQ to TAX. No significant differences ($p > 0.05$) were observed in the travel times between routes for all modes of transport, which allows route traffic effects to be discarded, and also allows eastward and westward journeys to be compared separately. Travel times ranged between 0.30 h for the trolleybus to 1.07 h for walking. All $PM_{2.5}$ time-series exhibited a log-normal distribution. Table 3 lists descriptive statistics for journeys from MAQ to TAX and from TAX to MAQ. Arithmetic averages (AAs) for $PM_{2.5}$ recorded from TAX to MAQ ranged from $16.5 \pm 6.5 \mu\text{g m}^{-3}$ for walking to $81.7 \pm 9.1 \mu\text{g m}^{-3}$ for bicycle, while respective medians ranged between 14.7 and $81.8 \mu\text{g m}^{-3}$. From MAQ to TAX, AAs of $PM_{2.5}$ ranged from $21.0 \pm 7.1 \mu\text{g m}^{-3}$ for walking to $75.3 \pm 9.3 \mu\text{g m}^{-3}$ for bicycle, whereas respective medians ranged from 20.7 to $78.1 \mu\text{g m}^{-3}$. No significant differences ($p > 0.05$) between geometric averages (GAs) and AAs were determined for all transport modes.

Table 3. $PM_{2.5}$ exposure concentrations in $\mu\text{g m}^{-3}$ by transport mode measured along MAQ Rd.

Mode of Transport	MAQ to TAX					
	Min *	Median	Arithmetic Average	Geometric Average	Max **	COY 1-h Average
Walking	11.0	20.7	21.0 ± 7.9	19.9	112.2	27
Bicycle	63.1	81.4	80.8 ± 9.3	80.5	102.9	50
Trolleybus	27.4	40.4	39.9 ± 6.4	39.4	54.7	45
CC	Discarded					
HEC	Discarded					
Mode of Transport	TAX to MAQ					
	Min *	Median	Arithmetic Average	Geometric Average	Max **	COY 1-h Average
Walking	9.1	14.7	16.5 ± 6.5	15.7	104.0	22
Bicycle	42.0	81.8	81.7 ± 9.1	81.1	163.7	43
Diesel bus	39.0	48.8	49.8 ± 5.3	49.5	71.3	32
CC	9.0	27.1	28.3 ± 8.5	27.7	111.9	42
HEC	11.0	19.8	20.3 ± 4.2	19.8	33.3	57

* Minimum; ** Maximum.

Figure 2 shows the box and whisker plots for $PM_{2.5}$ exposure concentrations recorded from TAX to MAQ and from MAQ to TAX by mode of transport, and the corresponding 1-h averages in $PM_{2.5}$

recorded simultaneously at the COY site. To account for the effect of PM_{2.5} emissions dispersion, WS and WD are also shown in Figure 2. Overall, no significant differences ($p > 0.05$) were observed among WS during the sampling campaign, while WD occurrence during the walking measurements was significantly different than the WD occurrence for the other modes of transport. The calculated PM_{2.5} residuals depend strongly on the mode of transport with the highest residuals of 31–39 $\mu\text{g m}^{-3}$ determined for bicycle. A lower residual of 18 $\mu\text{g m}^{-3}$ was determined for the diesel bus, while walking and the trolleybus exhibited negative residuals of 5–6 $\mu\text{g m}^{-3}$. This suggests that the exposure levels for walking and trolleybus are similar to those observed within the local area, while exposure while cycling and taking the diesel bus is significantly higher than that in the local area. By contrast, the CC and HEC showed negative residuals of 14 and 36 $\mu\text{g m}^{-3}$, respectively, which suggests a significant decrease in the personal exposure to PM_{2.5} compared to the other modes of transport studied.

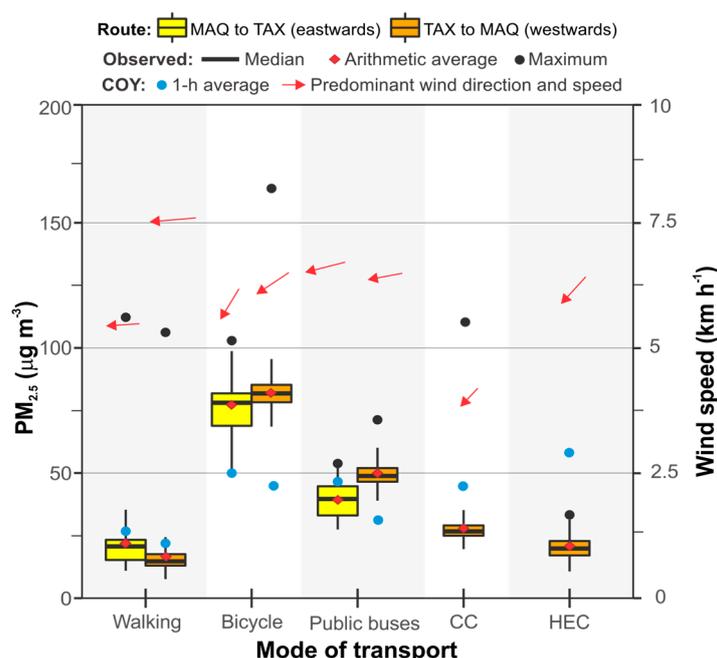


Figure 2. Boxplot of PM_{2.5} data recorded along the MAQ Rd in southern MCMA by mode of transport and route during the sampling campaign of November 2014. The eastward public bus journey was made on a trolleybus (yellow) and the westward journey on a diesel bus (orange). Red arrows show wind angle scaled to wind speed i.e., the longer the arrow, the higher the wind speed, as recorded at the COY site.

3.2. Spatial Variations in Street-Level PM_{2.5} Concentrations

Figures 3 and 4 show the spatial variations in PM_{2.5} observed for the eastwards and westwards routes along MAQ Rd, respectively, with peaks representing hotspots in personal exposure to PM_{2.5}. Horizontal lines in each figure represent 1-h averages of PM_{2.5} recorded at COY and are shown as reference for the street-level measurements. Overall, walking and trolleybus exhibit exposure levels to PM_{2.5} below those recorded at COY for most of the journey, while the bicycle shows levels higher than at COY for the whole journey. The peaks in PM_{2.5} observed from MAQ to TAX for all transport modes show large spatial heterogeneity, with the largest peaks observed for walking and the smallest for trolleybus (Figure 3a). Bicycle and walking show higher PM_{2.5} concentration peaks from MAQ to TAX than the trolleybus, which coincided with 4-way intersections and was not observed for the trolleybus (Figure 3b). Such peaks in concentrations of PM_{2.5} could arise from combined emissions of PM_{2.5} from vehicles accumulating and waiting in front of traffic lights [18]. The greatest peak in PM_{2.5} observed coincided with a large number of vehicles queuing to cross Division del Norte Rd.

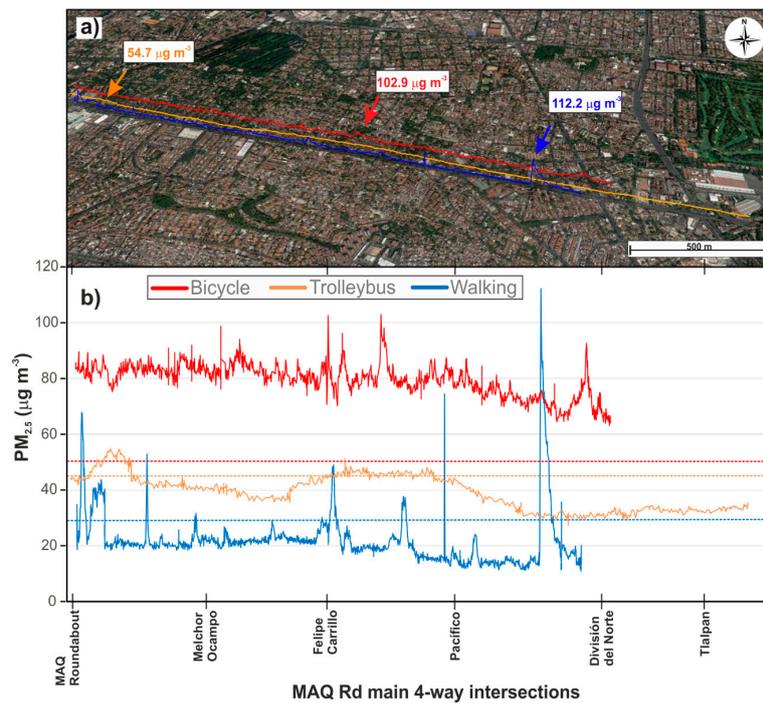


Figure 3. (a) Spatial location of PM_{2.5} by mode of transport recorded during MAQ to TAX (Taxqueña) journeys; (b) PM_{2.5} time series per journey. Street names indicated in the X-axis correspond to 4-way intersections. The horizontal dashed lines show corresponding 1-h averages of PM_{2.5} recorded at COY used to calculate exposure residuals.

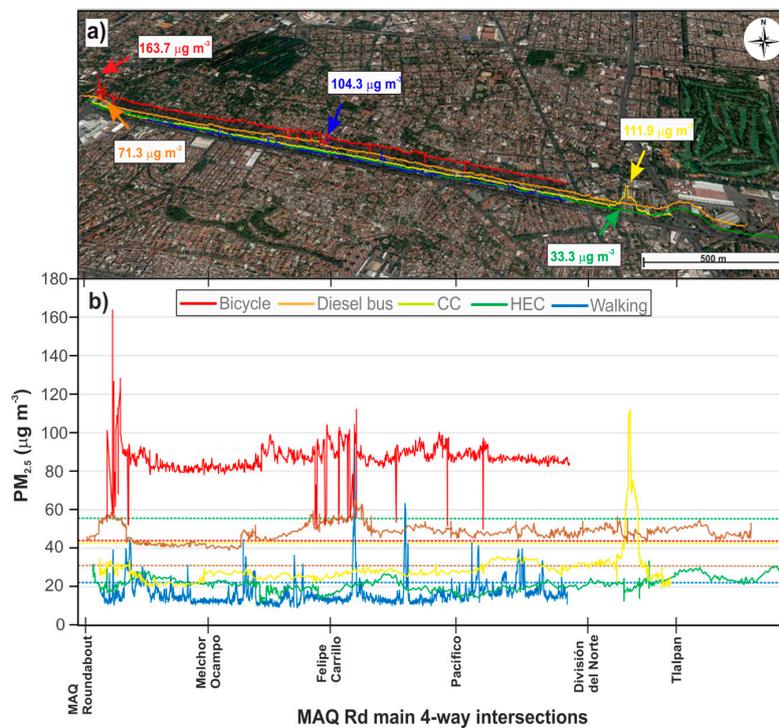


Figure 4. (a) Spatial location of PM_{2.5} by mode of transport recorded during TAX to MAQ journeys; (b) PM_{2.5} time series per journey. Street names indicated in the X-axis correspond to 4-way intersections. The horizontal dashed lines show corresponding 1-h averages of PM_{2.5} recorded at COY used to calculate exposure residuals.

Figure 4a shows peaks in $PM_{2.5}$ observed during journeys made from TAX to MAQ, which exhibited less spatial heterogeneity than those observed for MAQ to TAX journeys. Only close to the MAQ roundabout did all modes of transport, apart from the HEC, exhibit noticeable increases in $PM_{2.5}$. However, as for the MAQ to TAX journeys, most of the peaks in $PM_{2.5}$ were observed close to 4-way intersections (Figure 4b). The large increase in $PM_{2.5}$ observed for the CC was originated by emissions from a large number of cars queuing to pick up children from a primary school close to the MAQ Rd and Virginia street corner. Clearly, the lowest variations of $PM_{2.5}$ exposure concentrations are observed for the diesel bus, CC and HEC, while the highest ones correspond to bicycle and walking in good agreement with data recorded from MAQ to TAX. This suggests that commuting in vehicles may protect commuters to significant increases in the exposure to $PM_{2.5}$.

3.3. $PM_{2.5}$ Inhalation

Estimated inhalation of $PM_{2.5}$ ranged from 6.3 ± 1.0 to $147.4 \pm 69.4 \mu\text{g}$ for the trolleybus and bicycle, respectively, from MAQ to TAX, whereas from TAX to MAQ, it ranged from 4.7 ± 1.2 to $104.5 \pm 49.1 \mu\text{g m}^{-3}$ for the HEC and bicycle, respectively (Table 4). Estimated inhalation of $PM_{2.5}$ from ground-based measurements from MAQ to TAX ranged between $7.1 \pm 1.1 \mu\text{g}$ for trolleybus and $91.3 \pm 43.0 \mu\text{g}$ for bicycle. From TAX to MAQ, ground-based calculated inhalation ranged between $8.2 \pm 1.3 \mu\text{g}$ for the diesel bus to $55.0 \pm 25.9 \mu\text{g}$ for the bicycle. Higher inhalation of $PM_{2.5}$ for bicycle and diesel bus was estimated from on-field measurements than from ground-based measurements. Table 4 shows $\mu\text{g km}^{-1}$ of $PM_{2.5}$ inhaled as function of the mode of transport, and relative to $PM_{2.5}$ concentrations recorded at the COY site. For both routes, the highest inhalation was determined for bicycle (30.1 ± 14.2 to $39.0 \pm 18.4 \mu\text{g km}^{-1}$), while the lowest one corresponded to the public buses (1.5 ± 0.2 to $3.0 \pm 0.5 \mu\text{g km}^{-1}$). For all journeys, the lowest inhalation of $1.0 \pm 0.3 \mu\text{g km}^{-1}$ corresponded to the HEC from TAX to MAQ.

Table 4. Distance, inhaled $PM_{2.5}$ and residual inhalation by route and mode of transport along MAQ Rd.

Mode of Transport	MAQ to TAX				
	On-Field Measurements		COY Measurements		Residual Inhalation * (%)
	$\mu\text{g Inhaled}$	$\mu\text{g Inhaled km}^{-1}$	$\mu\text{g Inhaled}$	$\mu\text{g Inhaled km}^{-1}$	
Walking	12.1 ± 1.9	3.5 ± 0.6	15.6 ± 2.5	4.5 ± 0.7	-28.6
Bicycle	147.4 ± 69.4	39.0 ± 18.4	91.3 ± 43.0	24.2 ± 11.4	37.9
Trolleybus	6.3 ± 1.0	1.5 ± 0.2	7.1 ± 1.1	1.7 ± 0.3	-13.3
CC	-	-	-	-	-
HEC	-	-	-	-	-
Mode of Transport	MAQ to TAX				
	On-Field Measurements		COY Measurements		Residual Inhalation * (%)
	$\mu\text{g Inhaled}$	$\mu\text{g Inhaled km}^{-1}$	$\mu\text{g Inhaled}$	$\mu\text{g Inhaled km}^{-1}$	
Walking	17.0 ± 3.4	5.3 ± 1.1	22.6 ± 4.5	7.0 ± 1.4	-32.1
Bicycle	104.5 ± 49.1	30.1 ± 14.2	55.0 ± 25.9	15.9 ± 7.5	47.2
Diesel bus	12.7 ± 1.9	3.0 ± 0.5	8.2 ± 1.3	1.9 ± 0.3	36.7
CC	9.4 ± 2.4	2.4 ± 0.6	13.8 ± 3.5	3.6 ± 0.9	-50.0
HEC	4.7 ± 1.2	1.0 ± 0.3	13.2 ± 3.4	2.9 ± 0.7	-190.1

* Estimated as on-field derived inhalation minus COY derived inhalation.

The levels of inhalation determined from ground-based measurements were consistent with those estimated from on-field $PM_{2.5}$ data. The highest inhalation of 15.9 ± 7.5 and $24.2 \pm 11.4 \mu\text{g km}^{-1}$ was estimated for bicycle, while the lowest ones of 1.7 ± 0.3 and $1.9 \pm 0.3 \mu\text{g km}^{-1}$ was determined for the public buses, with ratios highest-to-lowest inhalation ranging from eight times from MAQ to TAX to 14 times from TAX to MAQ. We observed the highest residual inhalation expressed as a percentage

for bicycle in both routes (37.9 to 47.2%), followed by the diesel bus (36.7%). By contrast, the lowest residual inhalations corresponded to the CC (−50.0%) and HEC (−190.1%) from TAX to MAQ. Only walking showed similar inhalation residuals in both routes (−28.6 and −32.1%). Within the study area, active transport modes, but mostly the bicycle, are associated with the highest inhalation of PM_{2.5}, while commuting by private transport may protect commuters from an increase in the exposure concentrations compared to those considered representative of local area levels.

4. Discussion

Personal exposure to PM_{2.5} has been typically assessed using air quality modelling and ground-based monitoring [2,18,19]. However, model performance can be affected by the quality of input data, i.e., significant variations in air pollutant levels recorded by ground-based monitoring, seasonality and proximity to large emissions sources. In particular, personal exposure to PM_{2.5} from vehicle emissions is highly distance-dependent and may vary significantly between few and hundreds of meters from roads [20,21]. The comparison between ground-based and on-field PM_{2.5} measurements can be used to improve personal exposure prediction and health effects. This study addressed personal exposure to PM_{2.5} in transport micro-environments using real-time measurements within the MCMA. Residual exposure and inhalation were defined to compare personal exposure to PM_{2.5} when commuting by two active modes (walking, bicycle) and three passive modes (public buses, CC and HEC).

4.1. Personal Exposure to PM_{2.5} in the MCMA

Assessments of personal exposure to PM_{2.5} in transport micro-environments have been conducted all over the world [22]. The median exposure levels reported here for public buses and CC are lower than those reported in the MCMA by Vallejo et al. [12] of 101.7 and 62.4 µg m^{−3}, respectively. However, the average exposure levels reported here are within the range of those reported by Okokon et al. [9] (14 ± 5 to 85 ± 39 µg m^{−3}) in three European cities for bicycle, public bus and car with open and closed windows. However, comparisons between on-field measurements and ground-based monitoring were not made. Average exposure levels of between 2.9 ± 2.3 µg m^{−3} and 11.0 ± 6.6 µg m^{−3}, lower than those reported here, were observed by Quiros et al. [17] for cycling, walking and driving with open and closed windows in a residential area of Santa Monica, California. We estimate that average exposure to PM_{2.5} for cyclists in our study is between 7.8 to 15.4 times higher than the average for cyclists in Santa Monica. The range of average exposure for commuters reported in this study is higher than that observed in Sacramento, California, by Ham et al. [23], who reported exposure concentrations between 5 and 15 µg m^{−3} for journeys made in public bus, private car and bicycle. Such differences are likely due to vehicles fitted with cleaner technologies in the USA. For instance, in California, diesel technology is regulated under the EPA 2010 standards [24], while in Mexico analogous standards have not been introduced yet. However, consistent with our results, cyclists exhibited the highest exposure among the transport modes studied. By contrast, exposure concentrations for CC and HEC when travelling with open windows in our study are around 50% lower than those observed in St. Louis, MO, USA, by Leavey et al. [25].

Our results also contrast with those reported by Adams et al. [1] from a multi-mode transport study carried out in Central London, who reported the highest geometric average exposure for public transport (bus, 38.9 µg m^{−3}), followed by CC (33.7 µg m^{−3}) and the lowest ones for bicycle (23.5 µg m^{−3}). We observed that for the same transport modes in the MCMA, the largest exposure occurs when cycling (81.1 µg m^{−3}), followed by the diesel bus (49.5 µg m^{−3}) and CC (27.7 µg m^{−3}). The differences in the exposure levels to PM_{2.5} suggest that each mode of transport can be influenced by different factors. For instance, the highest exposure observed for cyclists along the MAQ Rd arises from proximity to the traffic flow, which consequently increases immediate exposure to vehicular emissions [23]. The lower exposure observed for walking is due to the distance between footpaths and road edges, which in the study area varies between 1 and 4 m [19]. The estimated exposure residuals

confirmed that proximity to roads and, in particular, to vehicular emissions influence the personal exposure in transport micro-environments. In this study, no significant effect of difference in WD (wind direction) occurrence was observed on the exposure residuals. For instance, walking, CC and HEC exhibited negative values under different WD occurrence, while bicycle, public buses and CC and HEC exhibited positive and negative values under similar wind occurrence. However, further measurements are needed to support our results under more variable WD occurrence.

We observed that commuting in private transport may protect commuters from direct exposure despite the proximity to PM_{2.5} emissions sources. This contrasts with the findings of Okokon et al. [9], who observed a higher exposure to PM_{2.5} for the CC with open windows than for public bus and bicycle and with Ham et al. [23] who reported similar exposure for private cars and public buses. This could be due to differences in the traffic flow but mostly to vehicular fleet technology; while in Europe around 50% of total light duty vehicles are diesel-powered [26]; in the MCMA around 80% of the light duty vehicle fleet is petrol-powered [27]. The higher exposure to PM_{2.5} for the diesel bus reported here than for the trolleybus could be related to recirculation of exhaust gases inside the bus cabin due to the lack of diesel particulate filters fitted. Asmi et al. [28] reported higher concentrations of PM_{2.5} inside older buses than inside new buses related to differences in bus-to-bus emissions, which could explain the difference reported here. The lower exposure observed in private cars relative to public transport could be due to tailpipe diesel exhaust recirculating to the cabin, which was observed in the California school bus program [29]. Overall, the residuals estimated for the trolleybus, CC and HEC confirmed a lower exposure than that estimated for the other modes of transport evaluated here.

4.2. The Spatial Distribution of Peaks in PM_{2.5}

Along MAQ Rd, the highest concentrations of PM_{2.5} recorded in each mode of transport were observed close to 4-way intersections. The most plausible explanation for such plumes of PM_{2.5} is large emissions from idling motor vehicles during the idling/acceleration cycle. Jazcilevich et al. [30] reported a net increase in PM₁₀ emissions of 3.1% from diesel vehicles in the MCMA during the stop/acceleration cycle compared to vehicles travelling at constant speed. This was clearly noticed when walking from MAQ to TAX and, for the CC journey from TAX to MAQ when lines of >10 cars were observed close to the corner of Virginia St. and MAQ Rd. Such plume corresponded to the highest concentrations of PM_{2.5} recorded for HEC and CC. However, the link between plumes and traffic density could be improved significantly by performing vehicle counts or conducting visual recording. Leavey et al. [25] observed that concentrations of PM_{2.5} in-cabin followed closely with outdoor levels when travelling with open windows, which may explain the peak observed for the CC. Our results contrast with observations made in California by Quiros et al. [17], who reported an increase in the levels of ultrafine particles but not in PM_{2.5} when walking close to a diesel bus during the accelerating phase after a complete stop. Such difference could be related mostly to the number of vehicles passing by the area when the PM_{2.5} plumes were intercepted and cleaner diesel technology in California, while in the MCMA most of the diesel vehicles are not fitted with particulate filters.

We did not observe a noticeable effect of WD and WS on the spatial variability of the PM_{2.5} peaks between routes for each mode of transport. Venkatram et al. [21] addressed the effect of WD on air pollutants dispersion in the USA reporting three scenarios, (i) little variation of concentrations with WD at downwind receptors; (ii) the highest concentrations recorded when WD was perpendicular to the Rd; and (iii) no significant changes in concentrations for a WD angle of 90 degrees. The spatial distributions of PM_{2.5} observed during the sampling campaign suggest no significant changes for an angle of WD between 0–100 degrees, which is in good agreement with the results reported by Venkatram et al. [21] for such a scenario. Moreover, the similar behaviour in the PM_{2.5} peaks observed for cycling and walking despite the differences in different WD and WS occurrence may confirm their non-significant effect on the exposure to peaks in PM_{2.5}. Our results agree well with those of modelling reported by Batterman et al. [19] in Detroit, U.S. of concentrations of pollutants decreasing

with distance from the road, as observed here for walking, and hotspots in $PM_{2.5}$ observed close to road intersections.

4.3. $PM_{2.5}$ Mass Inhalation

Mass inhalation of $PM_{2.5}$ by distance travelled (μg inhaled km^{-1}) calculated in this study are compared with that estimated for European and USA urban areas to put the inhalation values estimated at the MCMA in context. Consistently with our results, Ham et al. [23] and Panis et al. [31] reported the highest inhalation of $PM_{2.5}$ for commuters by bicycle compared to public bus and private car in three cities of Belgium and Sacramento, USA, respectively. However, inhalation for cyclists in our study exceeds between 14.3 to 18.5 times and 7.5 to 8.5 times that reported in Sacramento and in Belgium, respectively. Similarly, inhalation for car commuters in the MCMA was higher between 3.5 to 4.8 times that reported for Belgium [31], and 7.5 times that in Sacramento where commuting in public bus results in lower inhalation between 4.3 to 8.6 times than in MCMA [23]. Although such studies used a different approach to estimate inhalation based on average exposure levels, compared to the average levels by exposure windows used in this study, the results are consistent and confirm that cyclists exhibit the highest exposure risk followed by public buses and private cars. Public transport emissions per person are lower than from private cars [27]; nevertheless, highly pollutant buses expose other cleaner modes of transport, such as cycling, beyond acceptable limits, thus discouraging its use and popularisation. This suggests that to protect cyclist commuters, policy makers should consider the implementation of cycling lanes away from motorised traffic and along roads with low traffic, while inhalation reduction of public transport commuters could be achieved by the introduction of cleaner diesel technologies currently available in the market.

5. Conclusions

We assessed personal exposure to $PM_{2.5}$ in transport micro-environments within the MCMA. Five modes of transport and two routes were assessed, with average personal exposure to $PM_{2.5}$ during a single commute ranging between $16.5 \mu\text{g m}^{-3}$ for walking and $81.7 \mu\text{g m}^{-3}$ for cycling. The maximum exposure concentrations were observed for cyclists while the lowest ones corresponded to walking and CC. For commuters in motor vehicles, we observed a lower inhalation of $PM_{2.5}$ per km travelled for private cars than for public buses. The higher inhalation of $PM_{2.5}$ aboard diesel buses was likely due to recirculation of exhaust emissions inside the cabin due to open windows. This result disagrees with those reported in the USA and Europe where public transport often circulates with operating air conditioning and with closed windows, unlike in the MCMA. For active commuters, the highest inhalation of $PM_{2.5}$ per km travelled was observed for cycling compared to walking. Estimated residuals revealed a significant increase in inhalation due to the interception of high $PM_{2.5}$ concentration peaks. The use of residuals allowed comparison of local area and traffic derived exposure and inhalation, however, the definition of a baseline based on filtering techniques could improve such a comparison.

The results reported in this study provide information on $PM_{2.5}$ exposure levels in transport micro-environments together with information on critical regions of high concentrations in $PM_{2.5}$. This information can be used by local authorities, in particular, to design and modify lines for cyclists in an effort to secure distances away from diesel emissions. For instance, cyclist inhalation could be significantly reduced by installing bicycle lanes near footpaths and allowing a parking lane between those and traffic lanes. Another alternative could be to designate cycle lanes on adjacent streets to main roads. In addition, the introduction of hybrid and electric vehicles such as trolleybuses seems to represent a feasible opportunity not only to reduce motor vehicle emissions, but to decrease exposure risk for cyclists and pedestrians. The results presented in this study are consistent with existing studies that reported the highest exposure risk for cyclists circulating near or with the traffic stream. Our experiments were carried out along a representative road, where different modes of transport

circulate. Continuous monitoring of exposure concentrations along other major roads would aid interpretation of the PM_{2.5} dynamics observed, especially on those with high traffic loading.

Supplementary Materials: The following are available online at <http://www.mdpi.com/2073-4433/9/2/57/s1>.

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