Characterization of Road Dust Emissions in Milan: Impact of Vehicle Fleet Speed

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ABSTRACT

Road dust suspension is an increasing concern in terms of being a source of atmospheric particulate matter (PM) in cities, due to its toxicity and the lack of knowledge on emission estimates, impact and mostly with respect to measures to control or mitigate it. Both technological and policy measures have been proposed, but their application is limited by the gap of knowledge on their effectiveness in the real world. This study analyses the real-world emission factors of road dust at ten sites in the city of Milan (Italy), with an emphasis on the impact of different fleet speeds at one particular road. PM_{10} emission factors were estimated by means of both the EPA method (based on silt loadings) and the vertical profile of dust deposition. Road dust silt loadings varied within 0.006–0.066 g m^{-2}, with the highest loadings found at sites affected by construction works. Typical urban roads were found to have fleet-averaged emission factors within 13–32 mg vehicle^{-1} km^{-1}, which is in the central range of the literature values in Europe. The emission factors estimated by means of the vertical profile approach were within 19–26 mg VKT^{-1}, which agree quite well with the EPA method if corrected for speed. In fact, a power-law relation was found between fleet speed and emission factor estimates, with an exponent close to 1.5 for speeds within 36–57 km h^{-1}. These results suggest that the limitation of the maximum traffic speed can be effective for mitigating road dust emissions in cities.

Keywords: Air quality; Resuspension; Measures; Policy; PM_{10}.

INTRODUCTION

Urban air pollution is a threat to public health: several studies demonstrated the increase of cardiovascular and respiratory hospitalization and premature deaths due to atmospheric particulate matter (PM) (Lim *et al.*, 2012; Stafoggia *et al.*, 2013). Road traffic is among the largest source of ambient air PM in cities. In spite of the large achievements in tail-pipe emission reductions due to the EUROx directives (AIRUSE, 2016), an important fraction of traffic emissions remain uncontrolled, namely non-exhaust emissions. Indeed, studies have demonstrated that the relative importance of non-exhaust emission is increasing, and is already becoming the dominant local source in cities (Amato *et al.*, 2014; Font and Fuller, 2016). Political and technological actions are therefore urgently needed, in order to regulate and minimize emissions from brakes, tires and road wear as well as from road dust resuspension. The reduction of the maximum allowed speed has been often

proposed as a measure to reduce PM emissions from traffic. The effect on air quality has been studied using monitoring and/or modelling data focusing on exhaust emissions, while fewer studies have been undertaken on road dust emissions.

Norman *et al.* (2016) and Hagen *et al.* (2005) studied the effect of vehicle speed reduction in a Norwegian highway in 2004–2006. Norman *et al.* (2016) applied an emission model to quantify the PM_{10} reduction, but the first year after the implementation was much drier than the previous, hampering the quantification of the effectiveness of the measure. Moreover, changes in other confounder factors such as traffic volume and the proportion of studded tires occurred.

Other studies have explored the variation of emission strength of a deposited tracer with the speed of one travelling vehicle, suggesting that reducing vehicle speed may significantly decrease road dust emissions (Sehmel, 1967; Nicholson and Branson, 1990), but no studies have demonstrated experimentally the effectiveness of such a measure in real conditions. Other studies were limited to the use of one single vehicle equipped with PM sensors to track emissions (Heinsohn *et al.*, 1975; Pirjola *et al.*, 2009, 2010; Lee *et al.*, 2013). Lee *et al.* (2013) found larger mean PM concentrations only when the vehicle speed increased from 80 to 110 km h^{-1}, but with high standard deviations

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(from 31.9 ± 31.2 to 40.5 ± 30.2). Pirjola *et al.* (2009) and (2010) tested in Finland and Sweden, respectively, the emissions at different speeds and with different tires. They found a linear increase of dust concentrations, measured behind the rear tire of two testing vehicles, exploring the range 50–80 km h^{-1}, but no emission factors (EFs, milligrams of coarse particulate matter [PM_{10}] per vehicle kilometer travelled [mg VKT^{-1}]) were calculated. Hussein *et al.* (2008) found an important dependence on vehicle speed in Stockholm (Sweden), being the particle mass concentrations behind a studded tire at 100 km h^{-1} about ten times higher than that at 20 km h^{-1}. However, these results might be only relevant to the case of studded tires. Etyemezian *et al.* (2003) and Zhu *et al.* (2009) found that, using non-studded tires under the same road conditions (i.e., same dust loading), emissions increase with vehicle speed to the power of approximately 3.

A higher number of studies have been performed on unpaved roads, where a strong exponential dependence of PM_{10} EFs on vehicle speed was found (Tsai and Chang, 2002; Gillies *et al.*, 2005; Kuhns *et al.*, 2010; Williams *et al.*, 2010; Kavouras *et al.*, 2016). Emission factors were found to increase exponentially from 16–32 km h^{-1}, but these results are relative only to one vehicle, and probably not applicable to paved roads.

No studies estimating the changes in emission factors for a typical Central-Southern European city are currently available and, most importantly, for real world conditions (actual fleet and real road dust reservoir). As a result of this lack of information, emission models have yet to implement vehicle speed as a control variable (Omstedt *et al.*, 2005; EPA, 2006; Kahuaniemi *et al.*, 2012), with the exception of that in Denby *et al.* (2013), where PM EF has a second power dependence on speed for water spray emissions with studded tires in the Nordic environment. However, they did not specify it for road dust suspension, stating that the model remains uncertain and requires further refinement based on experimental studies.

In this study, we have experimentally investigated the variability of road dust EFs as a function of average fleet speed in a outskirt road of the city of Milan (Italy), characterized by an increasing speed of travelling vehicles due to the presence of a traffic light. Speed varied from 0 to 60 km h^{-1} within a distance of 160 m. We have also quantified emission factors by means of two different methods at six other contrasting roads.

STUDY AREA AND METHODOLOGY

Milan is the second most populated Italian city and one of the most populated in the European Union (Eurostat, 2014). As in many other areas of Northern Italy, the EU air quality standards for PM_{10} are not achieved in Milan: the high mountain chains that surround the Po River valley reduce the circulation of air masses, favouring the stagnation of pollutants and also the formation of secondary aerosols. In this scenario, evaluating the effectiveness of technological and non-technological measures is an important action, mostly for those aimed at the reduction of primary PM

emissions (Bedogni *et al.*, 2011).

According to the Lombardy Region emission inventory (INEMAR ARPA Lombardia, 2016), the most important source of primary PM_{10}s in the city of Milan is road traffic (44%), albeit emissions from road dust resuspension are not yet considered in the emission inventory. According to the Urban Mobility Plan of Milan, following the common European emission inventory guidelines (EEA, 2013), non-exhaust components (tire wear, brake wear and road abrasion) are estimated to be the most important fraction of the road traffic primary PM_{10} emissions for several years already, due to the emission reduction achievement from EU regulations on particulate exhaust emissions. The contribution of road dust resuspension could not be evaluated due to the lack of good quality local information (EFs), but air quality data records suggest that the importance of this phenomenon is increasing in Milan.

For these reasons, a research study was established in order to obtain real-world road dust emission factors in Milan. Also, the role of traffic vehicle speed has been quantified and evaluated.

Three sampling campaigns were performed. The first campaign, performed on 8th–9th September 2015 (average T: 20.7°C, average RH: 45%, no rain), allowed the collection of 12 samples of road dust silt loadings and the calculation of emission factors according to the EPA AP-42 model. The second campaign, performed from 8th September to 1st October 2015 (average: T 19.1°C, average RH: 57%. total rain: 70–71 mm), allowed estimation of EFs according to the vertical profile method (Amato *et al.*, 2012a) at three roads. The third campaign, also performed with vertical profiles from 20th May to 16th June 2016 (average T: 19.2°C, average RH 66%, total rain 201 mm), allowed estimating EF variability with vehicle fleet speed.

For the first campaign, six roads were selected in the city of Milan, namely Manin road (MAN), Venezia road (VE), Indipendenza road (IN), Liberazione road (LI), Mecenate road (ME) and Sarca road (SA) (Fig. 1). MAN and VE are within the central Restricted Traffic Area (RTA), where the circulation of vehicles with a length exceeding 7.5 meters is forbidden and controlled by an automatic number plate recognition system. IN and LI sites are located within the urban intermediate area, where the circulation of articulated lorries is forbidden and they were both influenced by construction works (Table 1). In particular, the IN site was located near an important construction site of a new underground line, while the LI site was affected by material lost by trucks of a smaller building site; both sampling sites were approximately 250–300 meters apart from the construction sites. At the last two locations (ME and SA) there are no limitations to traffic fleet. At each location, total deposited dust was collected from a half square meter (100 cm × 50 cm) of road surface (being the longer side perpendicular to the road axis and centered in the most right hand active lane) by means of a Becker vacuum pump at a 40 L min^{-1} flow rate on a 47 mm diameter quartz membrane filter (Pallflex TISSUEQuartz 2500QAT-UP) during 15 minutes. The filter was placed inside an in-line PVC filter holder with a collection chamber where coarser
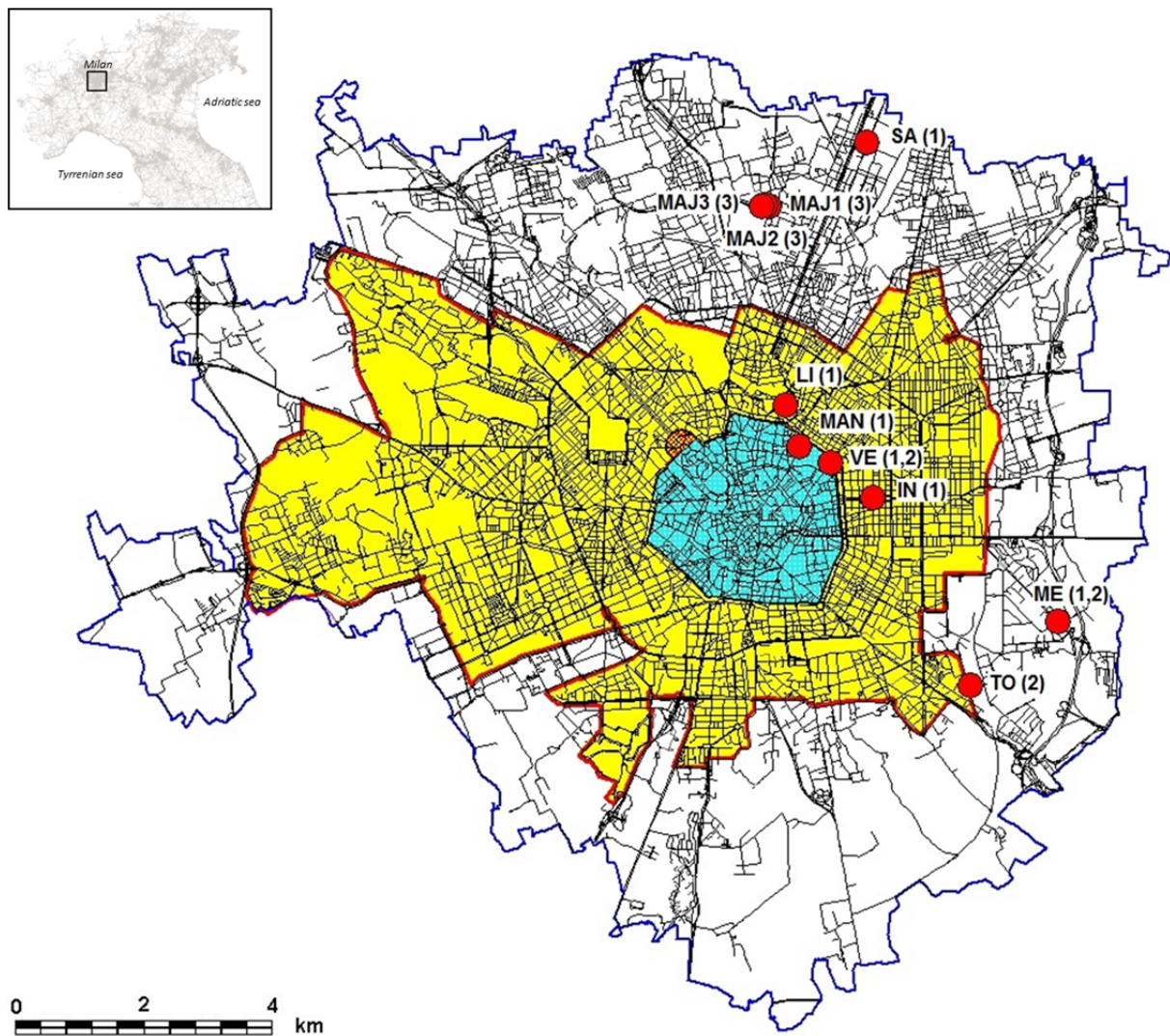
Fig. 1. Sampling sites across the city of Milan. Numbers in brackets identify the sampling campaigns. Yellow and blue areas correspond to the Intermediate and Restricted Traffic Areas (RTA), respectively.

Table 1. Details of silt loading sampling sites and values for silt loadings and Emission Factors (EF), as estimated by means of the EPA AP42 method. na: not available.

Site name	Type of road	Vehicle fleet speed (km h^{-1})	Mean vehicle weight (tons)	Silt loading (g m^{-2})	Mean Silt loading (g m^{-2})	EF (mg VKT^{-1})	Mean EF (mg VKT^{-1})
ME 1	Outskirts	46	2.3	0.006	0.00556	13.5	12.9
ME 2	Outskirts	46	2.3	0.005		12.2	
IN 1	Intermediate area; works nearby	na	1.9	0.064	0.06557	97.2	100.0
IN 2	Intermediate area; works nearby	na	1.9	0.068		102.7	
LI 1	Intermediate area; works nearby	na	1.9	0.032	0.04157	52.4	66.0
LI 2	Intermediate area; works nearby	na	1.9	0.051		79.4	
MAN 1	City Centre	na	1.4	0.009	0.01775	11.6	22.3
MAN 2	City Centre	na	1.4	0.027		32.5	
VE 1	City Centre Unpaved park influence	na	1.4	0.024	0.03048	29.3	36.5
VE 2	City Centre Unpaved park influence	na	1.4	0.037		43.5	
SA 1	Outskirts	50	1.9	0.017	0.01876	28.7	32.0
SA 2	Outskirts	50	1.9	0.021		35.3	

particles were kept. This sampling procedure was repeated twice on two adjacent 0.5 m^2 areas and on two different filters. After sampling, filters were kept in PETRI holders while bulk sediments from filter holders were stored in polyethylene sampling bags and brought to the laboratory to separate the fraction below 75 µm by means of a nylon sieve. The sieved samples were weighed with a Sartorius LA 130 S-F microbalance (1 µg sensitivity) to quantify the silt loading (g m^{-2}). Road dust EFs were estimated according to the formula proposed by the EPA (2006) in their AP-42 document:

$$EF = k\,(sL)^{0.91}\,(W)^{1.02} \tag{1}$$

where:
- EF = particulate emission factor (milligrams per vehicle kilometer travelled, mg VKT^{-1}),
- k = particle size multiplier for particle size range and units of interest (0.62 g VKT^{-1} for PM$_{10}$),
- sL = road surface silt loading (milligrams per square meter) (g m^{-2}), and
- W = average weight (tons) of the vehicles traveling the road.

For the second and third campaigns, deposited PM loadings were investigated as a function of height, using the samplers utilized by Amato *et al.* (2012a, 2016) for estimating emission factors from vertical profiles of dust deposition (below 10 µm) in the cities of Barcelona (Spain) and Paris (France). Sampler design details can be found in our previous publications (Amato *et al.*, 2012a, 2016); briefly, it consists of 15 cylindrical canisters of 6 cm height each and 7 cm of inner diameter. The canisters were installed in three vertical arrays, with canisters being separated by 1 cm from each other. Samplers were installed at the kerbside, at the smallest possible distance from the edge of the road.

The functioning of the samplers is similar to those used by Wagner and Leith (2000). Once exposed passively to ambient air PM deposition during 2–3 weeks, it is possible to obtain the PM vertical deposition profile for particles below 10 µm, as a result of two transport processes: deposition and the opposite rising movement of the particles due to turbulence. The asymptote of the exponential vertical profile is defined as terminal concentration m_∞ (unknown), which can be divided by the deposition area A, and the sampling time t (dry hours) in order to obtain the flux of deposition J:

$$J = \frac{m_\infty}{A \cdot t} \tag{2}$$

where m_∞ is estimated through a least squares fit to the equation:

$$\frac{m - m_\infty}{m_0 - m_\infty} = \exp\left(\frac{v_d}{D}\,z\right) \tag{3}$$

where z is the height, and m is the concentration at height z, v_d/D and m_0 (unknowns) and respectively the ratio between settling speed, and an effective particle eddy diffusivity, and the concentration at height 0, respectively. For the estimate of t, meteorological data were collected from the official database of the Lombardy Regional Environmental Protection Agency (www.arpalombardia.it).

The theoretical basis on which Eq. (3) is based can be found in Amato *et al.* (2012a), which assumes that the dry deposition of particulate matter is governed by diffusion- and gravitational-like mechanisms (Pasquill, 1962). Once the net deposition flux is obtained for particles with diameter below 10 µm, this can be converted to emission rate (Escrig *et al.*, 2011). If the road dust emissions depend on the amount of deposited dust in the road surface, then necessarily this dust loading has to evolve with time, until it is replenished at the same rate that it is removed. A simple mass balance helps to visualize this fact. Let M be the mass of road dust per unit area at a time t, and J the dust deposition flux. Supposing that the removal (i.e., emission flux) of road dust is a continuous process described by an unknown function of M, denoted by $f(M)$, the time evolution of M must obey:

$$\frac{dM}{dt} = J - f(M) \tag{4}$$

We will assume two additional hypotheses: (i) J is constant in time, and (ii) $f(M)$ increases monotonically with M. Indeed, these hypotheses are often tacitly assumed.

Suppose that at a certain instant the value of M is such that $f(M) < J$. Then, in the next (infinitely closer) instant, M will increase and the same will happen with $f(M)$, and therefore the difference $J - f(M)$ will become smaller. In an infinite succession of such events, $f(M)$ will progressively augment as its magnitude becomes balanced with J, and dM/dt approaches zero. As a consequence, given a long enough time, an equilibrium value of M will be reached.

This approach was previously followed by Nicholson (2001), who estimated the PM$_{10}$ resuspension by traffic in the UK from the assessment of its deposition. Nicholson's (2001) estimates are utilized in the UK emissions inventory (Murrells *et al.*, 2011).

Some studies also showed the empirical evidence of this process (Grottker, 1987; Amato *et al.*, 2012b). They monitored the time variation of M after a runoff episode, observing the asymptotic increase to a maximum value, corresponding to the equilibrium point between deposition and resuspension of particles.

A total of six arrays were installed during the two campaigns, at four different roads of Milan. In the second campaign, three arrays were installed (one each in ME road, VE road and in Toffetti road (TO)), while in the third campaign three samplers were mounted in Majorana road (MAJ)). These roads have contrasting traffic intensity and fleet composition, according to recent information available in Milan by means of the optical devices registering vehicle plates (Project Automation SpA - K53700 System). The three samplers installed at the same road (MAJ-X, MAJ-Y and MAJ-Z) allowed measuring the variability of road dust EFs as a function of average vehicles speed. MAJ is a two-way

road characterized by relatively high traffic volume (9,000–12,000 vehicles day km h^{-1} in each direction) and its two three-lined carriageways are separated by a large traffic island (19 m wide). Moreover, on-road parking is forbidden, thus the collection of road dust was not perturbed by parked vehicles. Finally, traffic lights compelled the majority of vehicles to accelerate gradually from 0 km h^{-1} up to 60 km h^{-1} at the last array (Fig. 2). The actual mean speeds at the three equipped sites have been estimated on the base of the analysis of the recorded data by a portable traffic camera (estimating for hundreds of vehicles the mean speed between two close virtual loops), some on-board speed measurements with one test car and almost one hundred travelling time measurements from one site to another, giving the following results: 36 ± 11 km h^{-1} at the first site, 47 ± 9 km h^{-1} at the second and 57 ± 9 km h^{-1} at the third site.

For each vertical profile, canisters were installed and removed after a time long-enough (at least 250 dry hours, but site-dependent) to obtain a minimum amount of sample (1–2 mg) in each canister for the laboratory analysis. After sampling, the canisters were brought to laboratory and the dust deposited in the canisters recovered with distilled water. The mass of dust was determined gravimetrically after sieving the collected dust at 250 μm and filtering the resulting suspensions with a Büchner flask into quartz and nitrate cellulose filters. Filters were dried at room temperature and conditioned for 48 h at 20°C and 50% relative humidity (RH). Weights were measured two times every 24 h by means of

a Sartorius LA 130 S-F microbalance (1 mg sensitivity). For each array, the sample from the fifth canister from ground canister was used for laser-based size distribution analysis (Malvern Mastersizer 2000 with Hydro 2000G), in order to study the particle size distribution (thus ensuring the road dust origin of collected particles) and quantify the fraction of mass below 10 μm, thus calculating emission factors for PM$_{10}$s.

RESULTS AND DISCUSSION

At each location, the silt loading (g m^2) was calculated as the average between the two sampled areas, multiplied by 2. Among the six locations, average silt loadings varied within 0.006–0.066 g m^{-2} (Table 1). The higher values were found for the roads affected by construction works – IN and LI. Moreover, the silt loading collected at the VE site was influenced by the road dust deposited by the service vehicles leaving the unpaved paths of the nearby public garden. Average values in roads not directly affected by external sources did not exceed 0.019 g m^{-2}, while such values in the VE site were 0.031 g m^{-2}, and values found at sites affected by urban construction works fall within the range of 0.042–0.066 g m^{-2}, confirming the heavy impact of construction works on road dust emissions.

Such values are in the lower range of those applicable to the EPA AP42 emission model, which span within 0.03–400 g m^{-2} (EPA, 2006), and this is likely due to the different
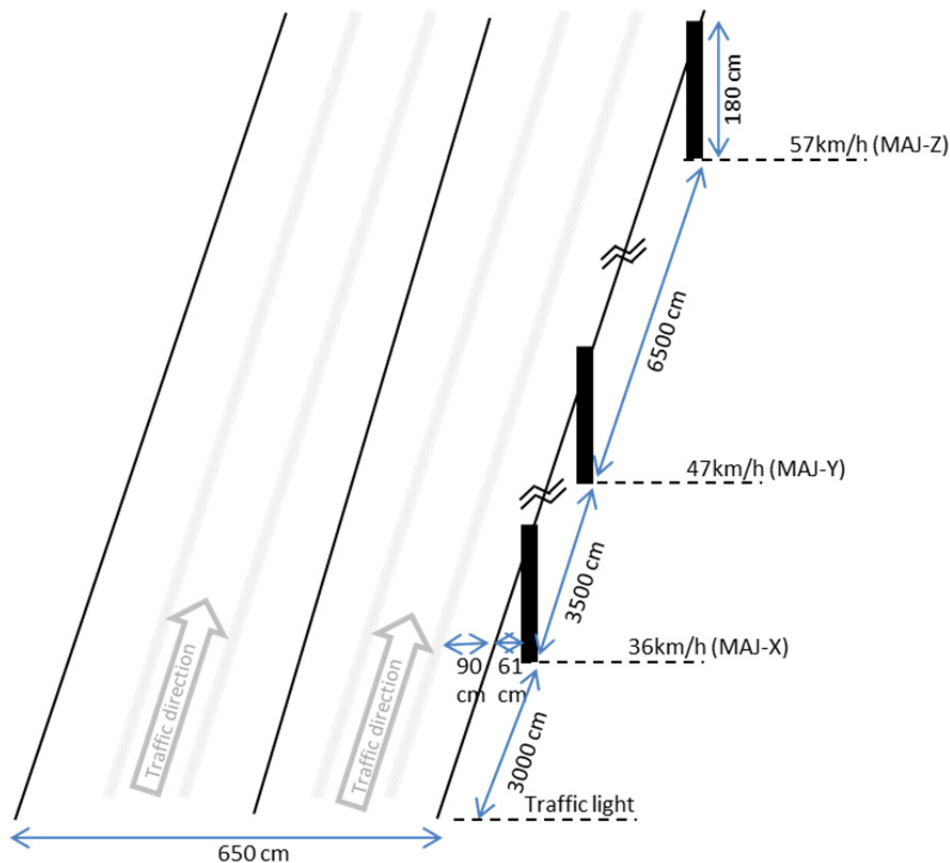

Fig. 2. Illustration of micro-scale vertical profiles (distances in cm) at MAJ road.

characteristics of roads between Europe and the USA. For W in Eq. (1), the mean vehicle weight was estimated based on local traffic count by means of the continuous recording of vehicle plates system of Milan. Mean emission factors for PM_{10} were estimated in the range 13–32 mg VKT^{-1} (excluding the locations clearly affected by external sources) depending on location and intended as fleet averaged, with an average value of 22.4 mg VKT^{-1}. Based on the limited literature available, the emission factors estimated in this study are well within the central range of the reported emission factors for PM_{10} at urban roads across Europe: Switzerland (1 ± 11 mg VKT^{-1}, only Light Duty Vehicles), UK (0–5 mg VKT^{-1}, only Light Duty Vehicles), Germany (57–109 mg VKT^{-1}), Spain (85 mg VKT^{-1}), Denmark (46–108 mg VKT^{-1}), Finland (121 mg VKT^{-1}) and Sweden (198 mg VKT^{-1}) (Ketzel *et al.*, 2007; Thorpe *et al.*, 2007; Amato *et al.*, 2010; Bukowiecki *et al.*, 2010). These estimates for the city of Milan confirm the general trend observed for Southern Europe, which experiences high solar radiation, favouring the mobility of deposited particles (Amato *et al.*, 2012b), as compared to Central and Western Europe, but no use of road sanding and studded tires (when compared to Scandinavian/Alpine regions), which are known to increase road dust loadings considerably (Kupiainen *et al.*, 2005). Concerning the roads affected by construction works or unpaved roads, the estimated EFs were considerably higher (37–100 mg VKT^{-1}) than for normal roads, thus increasing considerably kerbside PM concentrations in the proximity of the works. The mean EF values calculated for ME and VE were intended to be compared with independent estimates carried out by means of the vertical profiles in the second campaign.

Particle volume size distribution results for all the vertical profiles are reported in Fig. 3. Results show a rather coarse size distribution, typical of mineral dusty materials with mode within 30 and 70 μm, therefore associated mostly to road dust resuspension. The fraction below 0.1, 1, 10 and 100 μm correspond (on average) to 0, 1.6, 15.4, 78.8% of

particle volume. At all sites, the deposited mass of particles below 10 μm was calculated multiplying the percentage by volume of particles below 10.02 μm by the collected mass at each canister. The fraction below 10 μm was 14.8% and 13.5% in volume for ME and TO sites of the collected mass, respectively (below 250 μm). Results for the VE site are not reported because, during the sampling period, canisters were moved by unknown persons from their original position, therefore the values obtained cannot be taken as valid and need to be discarded.

The obtained mass (mg) of particles below 10 μm at different heights during the second campaign are plotted in Fig. 4, and were fitted to an equation of the form of Eq. (3) by means of a least squares method, solved by the Solver Excel tool. Concentrations diminish exponentially with height (x axis in Fig. 4). The shape of this profile is considered to be derived from the opposite transport processes of deposition and resuspension. Apart from the downwards deposition of PM, the traffic generates turbulence that results in an ascending flow of PM. Both mechanisms are, of course, transient. However, we will suppose that, on average, the system can be described by the previously introduced steady-state model (Eq. (4)).

The terminal concentration value obtained at each site can be used for calculating the deposition flux onto the surface ($z = 0$; Eq. (2)). The average terminal concentration m_∞ establishes the total emissions from traffic resuspension during the whole period of passive collection. However some contribution form direct wear might also be present, and could be estimated by source apportionment after chemical characterization. The fitting parameters and the calculated PM_{10} emission factors for each experiment are given in Table 2.

Very good correlation coefficients were found for all three sites, (R^2 = 0.96–0.98) indicating the validity of assumptions. EF estimates are based only on the hours (t in Eq. (2)) with dry road surface within sampling periods. In order to account for road surface moisture, we used the
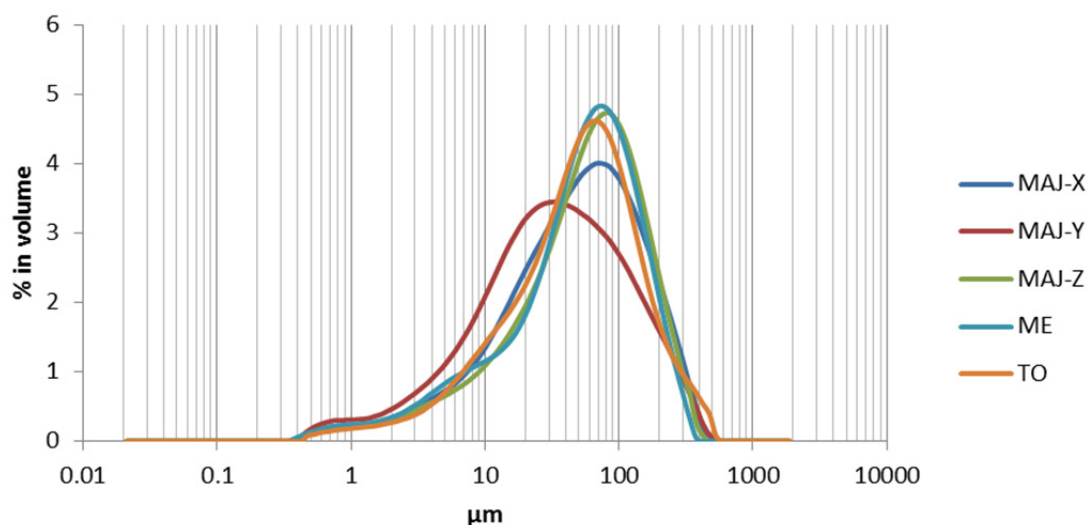
Fig. 3. Size distribution of samples collected at ME, TO and MAJ sites and sieved at 250 μm. For each array, the fifth lowest canister was used for the size distribution analysis.
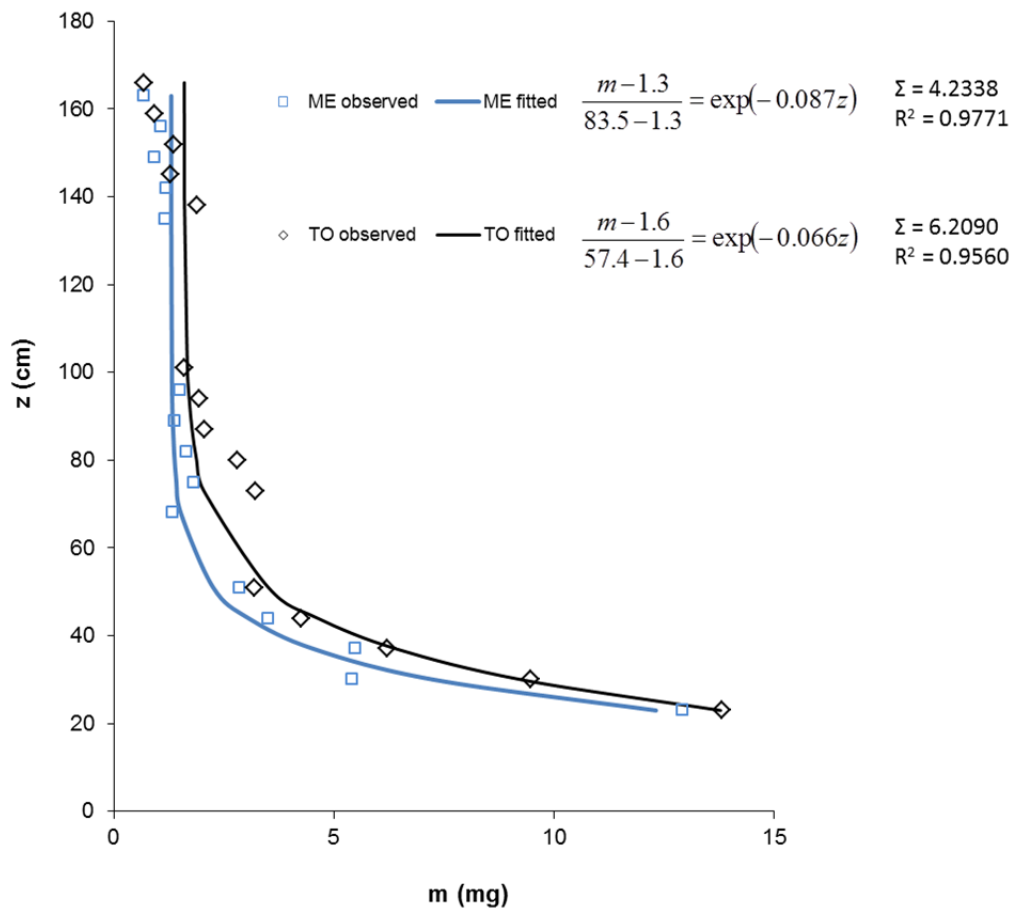
Fig. 4. Micro-scale vertical profiles of deposited particles (< 10 μm) concentrations at VE and ME sites. Σ is the sum of squared residuals.

Table 2. Emission factors for road dust in PM_{10} (EF_{10}) estimated by means of the micro-scale vertical profiles. na: not available.

	ME	TO
Vehicle fleet speed (km h⁻¹)	46	na
Total hours of exposure	558	574
Rainy hours	34	34
t (hours)	445	461
Cumulative amount of rain (mm)	70	71
Distance from the curb (m)	4	2
A (m²)	0.0034	0.0034
m_0 (mg)	83.5	57.4
m_∞ (mg)	1.3	1.6
v_d/D (m⁻¹)	0.09	0.07
R^2	0.96	0.98
J_{10} (mg m⁻² h⁻¹)	0.9	1.0
EF_{10} (mg VKT⁻¹)	**19.2**	**26.2**

estimate for Barcelona from Amato *et al.* (2012b), who found that 99% of road dust emission potential is reached 24 hours after the last rain event in Barcelona (Spain), whilst being 72 h for Utrecht (Netherlands). The time threshold of Barcelona was preferred since in Milan, during summer (May–September), the average annual temperature and relative humidity are respectively 19.0°C and 74%,

very similar to those in Barcelona (20.8°C and 72%), while considerably different than in Utrecht (14.9°C and 86%). We therefore discounted from the total number of sampling hours (558–574 hours) the hours with rain and those within 24 hours from the last rain event (113 hours).

The PM_{10} EFs estimated by means of the vertical profile approach were 19.2 and 26.2 mg VKT⁻¹ for ME and TO roads, respectively. These estimates must be intended as fleet-averaged. As already shown in Table 1, the average traffic speed was 46 km h⁻¹ for ME road (not available for TO road). The EF estimated at ME by means of the vertical profile was then 32% higher than what found with the EPA AP42 approach; however, this difference seems relatively low, considering that the two methods are based on different theories, assumptions, sampling and analytical techniques. The comparison is in fact satisfactory and it can be concluded that for the ME site, the overall estimate is 16. ± 4.5 mg VKT⁻¹. The higher EF found at TO site might be due to a different composition or a poorer state of the pavement. In both roads, the EFs found fall within the range 13–32 mg VKT⁻¹ estimated by means of the EPA AP42 method (Table 1). Summarizing, considering all sites not influenced by external sources (ME, TO, MAN and SA) and both methodologies, the overall estimate of the mean EF for the dry roads of Milan is 22.4 ± 9.5 mg VKT⁻¹.

In the third campaign, three arrays were installed in

MAJ road, as outlined in Fig. 2. The fraction below 10 µm was 14.4, 21.5 and 12.7% in volume for MAJ-X, MAJ-Y, and MAJ-Z sites of the collected mass (below 250 µm). Obtained masses (mg) below 10 microns at different heights are plotted in Fig. 5. Good correlation coefficients were found for all three sites, ($R^2 = 0.79$–0.97). Also in this case, emission estimates are based only on the hours after 24 h from last rain event, assuming the same hypothesis as for ME and TO sites. We therefore discount from the total number of sampling hours (644 hours) the hours with rain and those within 24 hours from the last precipitation (374 hours).

At MAJ sites, the fleet averaged PM_{10} EFs, estimated by means of the vertical profile approach, were 24.6, 40.9 and 48.4 mg VKT^{-1} for instantaneous traffic speeds of 36, 47 and 57 km h^{-1}, respectively (Table 3). The EF estimated at 36 km h^{-1} is similar to the mean value estimated for the other sites not influenced by external sources (22.4 ± 9.5 mg VKT^{-1}). Considering the sites at MAJ road with higher speeds, we can assume that any difference observed can be attributed to the change in vehicle speed, as samples were made under the same meteorology, road geometry, traffic intensity and fleet composition.

Besides the increasing gradient, a good correlation (Fig. 6) between traffic speed and EFs ($r^2 = 0.96$) was also found: emission factor increases as a power of traffic speed with an exponent close to 1.5. Fig. 6 shows an interval of emission factor values for each speed, where our best estimate is the lowest value while the (higher) extreme case is where emissions recover 72 h after rain. The 1.5 exponent is somewhat lower than that reported by Sehmel (1973), who found that resuspension increased with the square of the car speed, but higher than the estimates of Nicholson and Branson (1990) who suggested only around 20% increase of emissions from 36 km h^{-1} to 57 km h^{-1}. The difference between these estimates can be due to several factors, such as the amount and age of road dust, type of road dust (tracers were used by the aforementioned studies), road pavement and type of vehicles, among others.

If the relation between resuspension and traffic speed is taken into account, it is reasonable to suppose that the EFs estimated at the other roads of Milan by means of the vertical profile approach were also affected by the local traffic speed.

The vertical profile at the ME site was installed just near a traffic control point able to automatically recognize the plate number and the instantaneous speed of the vehicles. The measured mean speed at ME site was 47 ± 15 km h^{-1}, thus very similar to the speed at the MAJ-Y site. The EF of the MAJ-Y site is two-fold higher in comparison to the ME site, probably because MAJ road passes through a large peri-urban park, and higher than usual wind speed was registered during the MAJ campaign. Thus, the silt loading at MAJ sites could be partially affected by the dust input coming from the park, similarly to the VE site, which
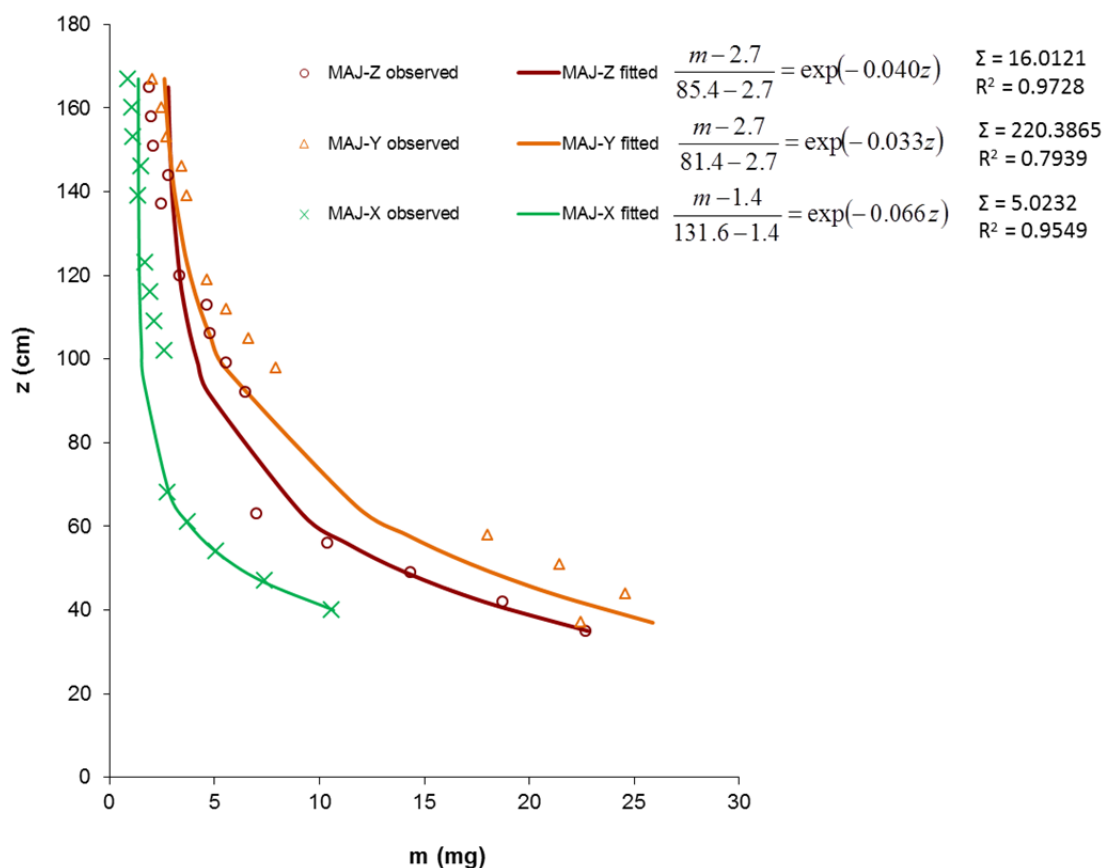

Fig. 5. Micro-scale vertical profiles of deposited particles (< 10 µm) concentrations at MAJ-X (36 km h^{-1}), MAJ-Y (47 km h^{-1}) and MAJ-Z (57 km h^{-1}) sites. Σ is the sum of squared residuals.

Table 3. EFs for road dust in PM_{10} estimated by means of the micro-scale vertical profiles at MAJ road.

	MAJ-X	MAJ-Y	MAJ-Z
Vehicle speed fleet	36	47	57
Total hours of exposure	644	644	644
Rainy hours	95	95	95
t (hours)	270	270	270
Cumulative amount of rain (mm)	201	201	201
Distance from the curb (m)	0.6	0.6	0.6
A (m^2)	0.0034	0.0034	0.0034
m_0 (mg)	131.6	81.4	85.4
m_∞ (mg)	1.4	2.3	2.7
v_d/D (m^{-1})	0.07	0.03	0.04
R^2	0.95	0.79	0.97
J_{10} (mg m^{-2} h^{-1})	1.5	2.5	3.0
EF_{10} (mg VKT^{-1})	**24.6**	**40.9**	**48.4**
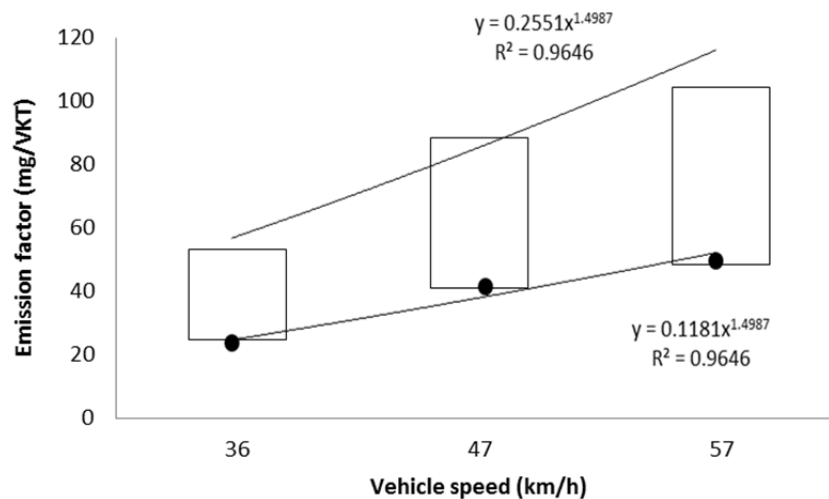
Fig. 6. Relation between resuspension emissions and average fleet speed (km h^{-1}) at MAJ road. Histograms represent the range of emission factors for recoveries between 24 and 72 hours after rain (Amato *et al.*, 2012b).

indeed shows an EF (36.5 mg VKT^{-1}) similar to MAJ-Y (40.9 mg VKT^{-1}). At the VE site, a direct measure of the mean traffic speed was not available, but it is estimated to be quite similar to the speed at the MAJ-Y site (within the range of 40–50 km h^{-1}).

As already shown, at the ME site the EF was estimated on the basis of two approaches: EPA AP42 (12.9 mg VKT^{-1}) and vertical profile (19.2 mg VKT^{-1}). The EPA AP42 approach does not take into account explicitly the traffic speed, but if we suppose that the "reference" speed for the EPA AP42 is 36 km h^{-1}, as average speed in urban areas, and correct it for 47 km h^{-1} with the relation found, the same value (19.2 g KT^{-1}) is reached at the ME site. Of course, this hypothesis is related to the range of the model applicability (speed below 88 km h^{-1}, according to the EPA AP42 method's range of application) and to the situation of Milan, in terms of atmospheric conditions, asphalt used, and traffic composition among others.

Our results demonstrate planning strategies for the reduction of the maximum traffic speed can help reducing PM emissions from road dust resuspension in urban area, which are also expected to increase in the long term due to the increasing number of heavier passenger cars such as electric and hybrid vehicles (in Milan, the number of hybrid cars is already increasing by 20–30% per year) (EPA, 2006; Timmers *et al.*, 2016).

CONCLUSIONS

Road dust silt loadings in Milan were found to vary within 0.006–0.066 g m^{-2}, as measured at six different roads in Milan's city center and peripheries. According to the EPA AP42 model, these silt loadings can be converted into average emission factors for each specific road. The range of emission factors in Milan was 13–100 mg VKT^{-1}, depending on location and intended as fleet averaged, with a total average value of 22.4 mg VKT^{-1} (considering roads not affected by construction works), while at the roads affected by construction works the estimated emission factors are considerably higher (66–100 mg VKT^{-1}). Such emission factors are generally lower than values found by other studies in Scandinavia and Alpine regions, where road sanding and studded tires can increase values, but relatively higher than in Central and Western Europe.

The alternative approach for emission factor estimates, performed by means of vertical profiles of dust concentrations, allowed estimating a similar value (32% higher) at one site (ME) and providing a range between 19 and 26 mg VKT^{-1} at two roads in Milan.

The impact of traffic speed on road dust emissions was investigated calculating emission factors at three sites of the same road with different traffic speeds. EF values were 24.6, 40.9 and 48.4 mg VKT^{-1} for traffic speeds of 36, 47 and 57 km h^{-1}, respectively, which suggests that road dust resuspension increases with a power of 1.5 of the vehicle speed.

Taking into account this power-law relation between resuspension emission factor and traffic speed, the EF values estimated at the ME site with the two approaches agree well if we suppose that the EPA AP42 model refers to a mean speed of 36 km h^{-1} and correct it for 47 km h^{-1}.

From the local policy point of view, our results suggest that urban strategies aimed to decrease the airborne particulate concentrations at local and microscales should consider, among others, the limitation of the maximum traffic speed and the reinforcement of the mitigation and control processes on the dust dispersed from construction sites.

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SUPPLEMENTARY MATERIAL

Supplementary data associated with this article can be found in the online version at http://www.aaqr.org.

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