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This is the author's manuscript

Original Citation:

Availability:

This version is available <http://hdl.handle.net/2318/1651749> since 2020-10-30T17:36:00Z

Published version:

DOI:10.1016/j.envpol.2017.10.115

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6 **This is the post-print version of the contribution published as:**

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Environmental Pollution 2017

<https://doi.org/10.1016/j.envpol.2017.10.115>

8

9 **The publisher's version is available at:**

10

11 <http://www.sciencedirect.com/science/article/pii/S0269749117332864>

12

13

14 **When citing, please refer to the published version.**

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18 **An empirical model to predict road dust emissions based on pavement and**
19 **traffic characteristics**

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48 **Abstract**

49 The relative impact of non-exhaust sources (i.e. road dust, tire wear, road wear and brake wear particles) on
50 urban air quality is increasing. Among them, road dust resuspension has generally the highest impact on PM
51 concentrations but its spatio-temporal variability has been rarely studied and modeled. Some recent studies
52 attempted to observe and describe the time-variability but, as it is driven by traffic and meteorology,
53 uncertainty remains on the seasonality of emissions. The knowledge gap on spatial variability is much wider,
54 as several factors have been pointed out as responsible for road dust build-up: pavement characteristics, traffic
55 intensity and speed, fleet composition, proximity to traffic lights, but also the presence of external sources.
56 However, no parameterization is available as a function of these variables.

57 We investigated mobile road dust smaller than 10 μm (MF10) in two cities with different climatic and traffic
58 conditions (Barcelona and Turin), to explore MF10 seasonal variability and the relationship between MF10
59 and site characteristics (pavement macrotexture, traffic intensity and proximity to braking zone). Moreover,
60 we provide the first estimates of emission factors in the Po Valley both in summer and winter conditions. Our
61 results showed a good inverse relationship between MF10 and macro-texture, traffic intensity and distance
62 from the nearest braking zone. We also found a clear seasonal effect of road dust emissions, with higher
63 emission in summer, likely due to the lower pavement moisture. These results allowed building a simple
64 empirical mode, predicting maximal dust loadings and, consequently, emission potential, based on the
65 aforementioned data. This model will need to be scaled for meteorological effect, using methods accounting
66 for weather and pavement moisture. This can significantly improve bottom-up emission inventory for spatial
67 allocation of emissions and air quality management, to select those roads with higher emissions for mitigation
68 measures.

69

70

71 **Keywords**

72 PM10; non-exhaust; resuspension; macrotexture; aggregate size

73

74 **Capsule**

75 A simple empirical model, predicting maximal resuspension emission potential, was developed basing on
76 pavement macrotexture, traffic intensity and proximity to braking zone.

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82 Introduction

83 Particulate matter (PM) emissions from road dust resuspension are an increasing concern for air quality and
84 public health (Denier van der Gon et al., 2013; Amato et al., 2014). The stricter PM emission standards adopted
85 in Europe, with the EURO Directive, have brought now more attention to the non-exhaust particles from road
86 traffic (i.e. road dust, tire wear, road wear and brake wear particles), for which emissions are not controlled,
87 and their relative impact on urban air quality is increasing (Amato et al., 2014b; Barmadipos et al., 2012). In
88 Southern Spain, for example, Amato et al. (2104b) found decreasing contributions for motor exhaust ($p <$
89 0.001) of 0.4 ($0.57-0.24$) $\mu\text{g m}^{-3}\text{year}^{-1}$ from 2004 to 2011. Conversely, in the same period, road dust
90 contributions to PM10 levels remained stable. Current estimates suggest that, in the European domain, non-
91 exhaust vehicle emissions represent 70% of urban primary PM10 emissions since 2015 (Keunen et al., 2014).
92 Among non-exhaust sources, road dust resuspension has generally the highest impact in PM concentrations,
93 however, a comprehensive assessment of road dust impact both in terms of pollutants (dust, carbonaceous
94 particles and metals) concentrations and health outcomes has been rare (Ostro et al., 2011). This is due to the
95 lack of reliable emission factors (for source oriented models, Schaap et al., 2009), and the similarity of the
96 chemical composition of road dust with other mineral sources (Amato et al., 2009). Moreover, experimental
97 studies on road dust characterization generally found high variability of road dust loadings in space and time,
98 suggesting the need of bottom-up inventories and improvement in the description of spatial and temporal
99 variability. Spatial inequalities of air pollution levels have been in fact observed at the urban scale, mostly for
100 PM10 and its coarser fraction, which is dominated by road dust resuspension. Time variability is crucial for
101 epidemiological studies but information is lacking on the day-to-day and seasonal variability of emission
102 factors. Some recent studies attempted to observe and possibly describe the spatio-temporal variability. In
103 Scandinavian countries, the use of studded tires and road salting/sanding are predictor variables to estimate
104 road dust loadings and the NORTRIP model has been successfully applied in order to predict both spatial and
105 temporal variability of road dust loadings (Denby et al., 2013a and 2013b); while in Central and Southern
106 Europe, where time-variability is driven by meteorology, in addition to traffic flow, several authors used the
107 ON/OFF approaches to account for road moisture due to precipitation (Pay et al., 2011; Schaap et al., 2009).
108 Amato et al. (2012) studied the day-to day variability of road dust suspendible fraction in one Spanish and one
109 Dutch streets, observing relatively short recovery rates within 24 h and 72 h respectively, which is in agreement
110 with the recovery of ambient air PM coarse curbside increment (Amato et al., 2011; Keuken et al.,
111 2010). However, uncertainty remains on the seasonal variability of road dust emissions, which can be also
112 affected by meteorological pattern (higher relative humidity in winter, higher drought in summer, intense
113 Saharan dust intrusions in spring, for example). Concerning the spatial variability, the gap of knowledge is
114 much wider. Several factors have been pointed out as responsible of road dust build-up (i.e. emission potential):
115 pavement characteristics (texture, mineralogy, age (Amato et al., 2013; Denby et al., 2013b; China and James,
116 2012; Berger and Denby, 2011; Gehrig et al., 2010; Gustafsson et al., 2009; Raisanen et al., 2005), traffic
117 intensity and speed, fleet composition (Bukowiecki et al., 2010; Liu et al., 2016), proximity to traffic lights,

118 but also the presence of external sources (e.g. construction dust, unpaved areas, African dust deposition, etc.).
119 However, no parameterization is available as a function of these variables. Moreover, to understand the impact
120 of these predictors is also important for air quality management since remediation measures can be designed.
121 In this study, we present a simple empirical model able to predict road dust suspendible fraction and,
122 consequently, emission potential, based on pavement macrotexture, traffic intensity and proximity to braking
123 zone. This model is based on field measurement in the cities of Turin (Italy) and Barcelona (Spain), thus
124 considering quite contrasting environments (Mediterranean and Continental) and aims to estimate the
125 maximum emission factor for single roads (without considering the effect of meteorology); it could be,
126 therefore, suitable for spatial allocation of emissions. Moreover, the article offers the first estimates of emission
127 factors in the Po Valley (Italy), one of the most polluted regions in Europe, both in summer and winter
128 conditions.

129

130 **Methods**

131 *Study area*

132 Four sampling campaigns were performed in the cities of Turin (Italy) and Barcelona (Spain), three in the
133 summer period (June and September 2016 in Turin, August 2016 in Barcelona) and one in winter (January
134 2017, in Turin). The two cities under study have a common high density of vehicle emissions (among the
135 highest in Europe) but different climatic conditions.

136 The Turin metropolitan area has a population of around 1.5 million inhabitants, and is the fourth most populous
137 metropolitan area in Italy. The high car density (5300 veh km⁻²) provokes (as sum of exhaust and non-exhaust)
138 almost 40% of the total primary emitted PM₁₀, the second most important source (after biomass burning),
139 according to the regional inventory (IREA, 2010). Climate is classified as humid subtropical, with moderately
140 cold but dry winters and hot summers, when rains are infrequent but heavy. Average rainfall is around 1000
141 mm per year (mostly concentrated in spring and autumn) and daily temperatures vary within 2-22 °C (monthly
142 averaged; ARPA Piemonte, 2014). The city is located at the western end of the Po valley, surrounded by hills
143 to the East and by the Alps to the North and West. Thus, the dispersion of pollutants is very low, as in all cities
144 of the Po valley (Eeftens et al., 2012; Belis et al., 2011), due to the very low wind speed and the thermal
145 inversion occurring in wintertime. Consequently, EC air quality standards are not met. In 2016, for example,
146 PM₁₀ concentrations in the city center exceeded the EU daily limit value for 70 days (ARPA Piemonte, 2016).
147 The Barcelona greater metropolitan area has around 4 million inhabitants and, with ~1 million vehicles, has
148 also one of the highest car densities in Europe (5900 veh km⁻²). The city is located in the western coast of the
149 Mediterranean basin, with a Mediterranean climate (mild winter and hot summers), and the scarce and
150 infrequent precipitation (614 mm as mean from 2010 to 2015) facilitate the mobilization and resuspension of
151 road-deposited particles. In Barcelona, traffic emissions are the most important pollution source of PM₁₀,
152 although other significant sources are mineral dust, shipping and industry (Amato et al., 2016). At urban
153 background sites, annual PM₁₀ concentrations due to the regional contribution are less than 30% of the total,
154 while non-exhaust vehicle emissions contribute 17% (7 mg m⁻³), with the total road traffic contribution

155 calculated at 46% (Amato et al., 2009). Thus, controlling local sources is very important for attaining PM10
156 limit values. The exposure scenario is even more problematic considering that 56% of urban population live
157 less than 70 m away from major roads (>10,000 vehicles day⁻¹).

158

159 *Road dust sampling*

160 The mobile fraction (able to be suspended under the applied airflow, 30 l min⁻¹) of road dust below <10 µm in
161 aerodynamic diameter (MF10) was sampled by means of a field dry resuspension chamber consisting in a
162 sampling tube, a methacrylate deposition chamber, an elutriation filter, where MF10 was separated, and a filter
163 holder, where particles were collected (a picture is available in Supplementary Material, Figure SM1). More
164 details on the sampler can be found in Amato et al. (2009; 2011).

165 In Turin, sampling sites were selected in order to characterize different fleet conditions, pavements and traffic
166 characteristics. To this aim, sites were chosen in residential, commercial and industrial neighborhoods and, at
167 some sites, sampling was performed in both traffic directions. In Barcelona, sampling sites were chosen in a
168 limited area, but on streets with different pavements and traffic conditions. In total, 72 filters were collected,
169 characterizing 30 sites in Turin and 6 sites in Barcelona (a map of sampling sites is reported in Figure SM2).

170 In order to ensure a complete re-establishment of the stationary conditions of dust loadings, all the samplings
171 were performed after, at least, one week without precipitation.

172 Each MF10 sample was collected as in previous studies, from a 50 x 100 cm area with the wider side centered
173 within the most-right active lane (excluding parking area), thus representing the area covered by wheel tracks
174 (Figure 1). For each site 2 filters were collected in adjacent areas to improve the representativeness of the final
175 road dust loading estimate.

176 Before sampling, quartz fiber filters (Pall, Ø47 mm) were dried at 205°C for 5 h and conditioned for 48 h at
177 20°C and 50% relative humidity. Blank filters were weighed three times every 24 h and kept in PETRI holders.
178 After sampling, filters were brought back to the laboratory and weighted after 24 and 48 h of conditioning at
179 the same T and HR condition.

180

181 *Pavement macro-texture measurements*

182 The macro-texture of road pavement corresponds to the size of the aggregate particles and the gaps between
183 them in the asphalt mixture (Henry, 2000). In our study, pavement macro-texture was characterized through
184 the combination of photographic analysis and the Mean Texture Depth (MTD) analysis (China and James,
185 2012). Pictures of the pavement surfaces were taken at a fixed distance using a 16.1 MP camera, with a
186 millimeter scale ruler in the field of view, and used to estimate average size of aggregates. In each site, two
187 photos were taken and the size distribution and mean horizontal size of aggregates was estimated by counting
188 the total number and the size of each aggregate along the ruler (Amato et al., 2013).

189 Since aggregates are embedded in the asphalt binder, the texture was estimated also by means of a MTD
190 analysis. This analysis, called sand patch method (China and James, 2012; Praticò and Vaiana, 2015), is a
191 standard evaluation of the surface macrostructure by careful application of a known volume of standard
192 material on the surface, and subsequent measurement of the total area covered (ASTM E 965, 2015). At each
193 site, two MTD measurements were made using 30 ml of glass spheres (clean silica sand passing a 300 µm and
194 retained on a 150 µm BS sieve, Mastrad ltd) and the average circular area was calculated using the average of
195 3 diameter measurements. This area was then divided by the applied volume of the spheres to calculate the
196 MTD.

197 The area used for photographic and MTD measurements was within the area of the MF10 measurement (Figure
198 1), and performed after the MF10 sampling and after sweeping the surface with a brush. Using these analyses,
199 we calculated the corrected aggregate mode (CAM) according to the formula proposed by China and James
200 (2012):

$$201 \quad (CAM)_i = MTD_i * (\text{aggregate mode})_i / (\text{aggregate mean})_i \quad (1)$$

202 where the CAM of the i-th sample is calculated starting from the MTD, the mode of the aggregates observed
203 through photographic analysis and the mean value of the horizontal size of the aggregates.

204

205 *Data analysis*

206 Multiple linear regression analysis was run to assess the feasibility of using pavement and site features to
207 predict MF10. As predictor variables, we used the corrected aggregate mode (CAM), the traffic intensity (TR)
208 and the distance (DIST) from the closest braking zone. The method was cross-validated using the leave-one-
209 out method (Todeschini, 1998). The indices used to interpret the validity of the model were the Relative
210 Prediction Error (RPE) the Standard Error of Cross-Validation (SECV) and the Cross-Validated Coefficient
211 of Determination (Q^2) (Gunawardena et al., 2014).

212 Regression curves have been calculated by means of a least squares fit, minimizing the sum of residuals scaled
213 by the standard deviation among duplicated measurements. All calculations were performed using Windows
214 Excel 2013 and IBM SPSS 23.

215 Data on traffic intensity was provided by the city council for Barcelona and, for Torino, by the in-house
216 company of the city (5T s.r.l.), refers to the monthly averaged daily fluxes in the sampled month. DIST was
217 calculated from the sampling site to the stop sign (or the traffic light line on the pavement).

218

219 **Results and Discussion**

220 *MF10 loadings and emission factors*

221 Descriptions of the sampling sites, macro-texture, traffic data, MF10 and emission factors for each site are
222 reported in Table 1. The MF10 loadings (the suspendible fraction of road dust) in Turin streets ranged between

223 0.8 – 42.7 mg m⁻², being the highest three registered at sites next to unpaved parks (12.3 mg m⁻²) and
224 construction works (11.5 and 42.7 mg m⁻²). Excluding these sites, the urban range was found to be 0.8-8.8 mg
225 m⁻² (mean of 2.0 mg m⁻²) which is in the central range of European cities characterized with the same
226 methodology (Amato et al., 2011; 2012; 2014, 2016; AIRUSE, 2016) but higher than other Central European
227 cities such as Paris (0.7-2.2), and Zurich (0.7-1.3). However, Turin measurements include summer and winter
228 data, while in Zurich and Paris measurements were only performed in winter. In fact, we observed a clear
229 difference between summer and winter samples (Figure 2), with a mean winter value of 1.0 mg m⁻²,
230 significantly lower than in summer (2.7 mg m⁻²). In Barcelona, summer MF10 loadings were found in the
231 range of 1.1 – 3.4 mg m⁻², confirming the values found in previous studies (Amato et al., 2012).

232 Road dust emission factors (EFs) were estimated based on MF10 using the power relationship proposed by
233 Amato et al. (2011), which has a similar theoretical basis of the USEPA AP-42 model but uses the MF10
234 instead of the silt (<75 µm) loading:

$$235 \quad EF_i [mg VKT] = a \cdot MF10_i^b \quad (2)$$

236 where MF10_i is the road dust suspendible fraction at the *i*th location, and *a* and *b* are empirically determined
237 coefficients. In the case of Barcelona sites, we used the coefficients (a=52.9; b=0.82) obtained in the same city
238 by Amato et al. (2012), while for Turin we used the average (a=45.9; b=0.81) between Barcelona and Zurich
239 due to the climatic conditions of Turin. Emission factors for each site are reported in Table 1 and must be
240 intended as fleet-averaged.

241 Typical urban roads in Turin showed annual-averaged values within 10 – 85 mg VKT⁻¹ (VKT: Vehicle
242 Kilometer Traveled, Table 1), with a mean value of 27±19 mg VKT⁻¹ (excluding the sites influenced by
243 construction works or unpaved areas nearby), which is the first estimate in the Po Valley and, to our knowledge,
244 the first annual-averaged value in Europe (accounting also for seasonal differences). Separating winter and
245 summer EFs the averages would be 20±8 and 31±16 mg VKT⁻¹, respectively, although not all sites were
246 sampled at the same spot both in summer and in winter (Figure 2). An explanation for this higher emission
247 factor in summer could be the lower pavement moisture, as all winter measurements were performed under
248 morning haze and high air humidity. In addition, road hygroscopicity could have been increased by road
249 salting, although only some roads were salted the week before the sampling campaign. Unfortunately, we do
250 not have any precise information on this operation (it was made at least two days before the sampling).
251 Moreover, other unknown dust sources, such as a higher soil contribution due to the drier climate, or
252 construction dust could have influenced the results.

253 In Barcelona, emission factors were estimated within 13 – 34 mg VKT⁻¹, with an average value of 20±8 mg
254 VKT⁻¹ (Table 1). If we compare our results with the emission factors estimated in European studies using the
255 same experimental set-up, these values are in the central (for Turin) or lower observed range. For example,
256 studies reported 5.4-17 mg VKT⁻¹ in Paris (Amato et al., 2016), 12-51 mg VKT⁻¹ in Switzerland (Amato et al.,
257 2012) and 77-480 mg VKT⁻¹ in southern Spain (Amato et al., 2013).

258

259 *Effect of traffic intensity and distance from braking zones*

260 The impact of traffic intensity on the road dust EF is poorly known although important for improving the
261 emission parameterization for modelling. Recent studies found or hypothesized lower EFs for high capacity
262 roads, as high traffic intensity and speed are both expected to lead to a lower road dust reservoir (Schaap et al.,
263 2009; Amato et al. 2013; Abu-Allaban et al., 2003; Etyemezian et al., 2003). Conversely, Bukowiecki et al.,
264 (2010) found the opposite, comparing the city center with a major freeway, revealing the need of more
265 empirical studies. In the present study, we have explored the relationship between MF10 and street
266 characteristics, such as traffic intensity and the distance from nearest braking zone (crossing, traffic light or
267 roundabout), using combined data of both cities to enhance the representativeness of the models. We used
268 traffic intensity data provided by city councils, based either on measurements or on mobility models. We
269 observed a rather good inverse correlation between MF10 and the number of vehicles circulating on the lane
270 (Figure 3), obtained by dividing the total traffic volume by the number of lanes. MF10 decreases with
271 increasing traffic intensity following a power law fit, reaching a background loading probably related to
272 pavement characteristics and meteorological conditions. In Figure 3, we have separated summer and winter
273 samples in order to avoid the meteorological disturbance on the goodness of the fitting. Below 1000 vehicles
274 day⁻¹ MF10 increased due to lower traffic volumes, that led to higher steady state MF10 than at high traffic
275 sites. However, also unknown sources could have contributed to the additional load, such as construction dust,
276 due to the poor state of the pavement (at some sites). Emission factors for secondary roads were therefore
277 higher than for high capacity roads. This finding is relevant for emission inventories, since using a single or
278 constant emission factor for the calculation of non-exhaust contribution to PM in a whole region (top-down
279 approach), may result in a mistake in emission estimates (Figure 4), e.g emissions from road with higher
280 intensity would be overestimated. The use of real or modeled EFs (as the site-specific relation in Figure 4) for
281 each road or area (bottom-up approach) for road dust emissions is therefore needed to improve modeling
282 results.

283 Brake wear particles are a major source of road dust (Grigoratos and Martini, 2015; Thorpe and Harrison,
284 2008). In Barcelona, Amato et al. (2013) estimated that 27% of MF10 was made of brake wear, as traced by
285 Cu, Sb, Ba among other elements. Therefore, we explored the relationship between MF10 and proximity to
286 braking zone (Figure 5) as a possible predictor of MF10 loading. We explored the relationship with the distance
287 both to the following braking zone and to the previous, or the smaller of the two. Results reported in Figure 5
288 show a rather poor correlation in the case of using the minimum distance but a negative trend is apparent,
289 suggesting that 70-100 m away from braking zone no significant increase in MF10 is observed due to brake
290 particles. These results are in agreement with Hagino et al. (2015), who found two peaks of brake PM emission
291 in dynamometer tests: the first peak during the application of a braking force and the second one afterwards,
292 during wheel rotation. In one case, the peak was even after 30 s of acceleration, thus after a considerable space
293 in the real world, and the emission during the acceleration phase represented almost 50% of the emitted PM.

294

295 *Effect of macro-texture*

296 In this study macro-texture was investigated through MTD and photographic measurements in order to
297 calculate CAM values. The classic MTD measurement typically resulted in a value indicating the mean size
298 of the gaps between aggregate particles, thus higher MTDs indicated deeper or larger pores (coarser texture),
299 while the CAM value also incorporated an estimation of the horizontal size of the particles.

300 MTD values ranged from 0.56 to 1.30 mm in Turin, and from 0.36 to 1.54 mm in Barcelona (Table 1), showing
301 quite a large variability in pavement macro-texture. Figure 6 shows the relationship between MF10 and both
302 the MTD and the CAM, using data of both cities but excluding roads with less than 1000 vehicles day⁻¹, where,
303 as described above, unknown sources could have contributed to the high loadings. In addition, in these streets
304 the traffic data was only modeled and not measured as in all other sites, thus introducing an additional error.
305 Results show a significant inverse correlation with both parameters ($r^2= 0.48$ and 0.54 for MTD and CAM,
306 respectively). The correlation is higher with the corrected aggregate mode (Figure 6), revealing that the higher
307 the porosity of asphalt, the higher the capability to inhibit resuspension. A similar result was found by China
308 and James (2012), who analyzed the relationship of mean corrected mode with the resuspension potential using
309 standard soil freshly applied to asphalt, and by Gehrig et al. (2010), using mobile load simulators. Moreover,
310 Jacobson and Wågberg (2007) found that, with studded tires, the aggregate diameter of the pavements
311 influences the total wear in as much as coarser material results in lower wear. However, those studies were
312 both dealing with controlled experiments, which might not be fully applicable to the real world, while our
313 study offers the first evidence, to our knowledge, in real-world conditions (regarding road dust loading and
314 physico-chemical properties). The resuspension inhibition could be influenced by the physical shielding of
315 pores and by the size distribution of particles in the bitumen matrix, as evidenced by the improvement of the
316 fit using the CAM. This improvement could be attributed to the use of the ratio of mode to mean (in equation
317 1). A value lower than 1 means that some large particles are enclosed in the matrix, so dust could be occluded
318 between these particles and the pavement or sheltered from them, causing a higher build-up or road dust. On
319 the other hand, if we have a number of particles finer than the average, they can smooth or occlude the pores
320 between larger particles, thus lowering the dust accumulation potential. Still, it must bear in mind that this
321 relationship is only valid in summer, being completely masked in winter by the stronger effect of road moisture
322 (Figure 6).

323 Recently, review studies on non-exhaust emissions (Denier van der Gon et al., 2013; Amato et al., 2014) have
324 suggested the possibility of the use asphalt with coarser texture to improve air quality by inhibiting road dust
325 resuspension, and our study offer experimental evidence, covering two different urban scenarios. Given that
326 the observed range of MTD variability is high (Table 1), even within the same urban area, there is space for
327 improvement since, currently, the surface texture is not regulated in cities. These results suggest that asphalts
328 that are more porous and with coarser textures could, therefore, be not only designed to reduce noise emissions
329 and to improve water drainage, but also to reduce the emissions from paved roads.

330

331 *Predicting model*

332 Our predicting model aims to estimate MF10 loading contributions from traffic sources, not from external
333 sources (e.g. construction dust, soil resuspension or African deposition). Therefore, only the spatial variability
334 could be assessed, as the temporal variability needs to be parametrized afterwards, using, for example, the
335 method proposed by Amato et al. (2012).

336 Based on the results and the relationships discussed above, a multiple linear regression analysis was developed
337 to predict the maximum MF10 loading from the CAM, the traffic intensity (TR) and the distance (DIST) from
338 the closest braking zone. Independently, each parameter showed an inverse relationship with the MF10, thus
339 we tried different linear regressions with two or three variables using exponential or potential equations. Our
340 results show that the equation with the best prediction of MF10 was:

$$341 \quad RD10 = e^{2.901 \pm 0.855} \times CAM^{-0.264 \pm 0.123} \times TR^{-0.218 \pm 0.105} \times DIST^{-0.145 \pm 0.076} \quad (3)$$

342 The model performed well, reporting the 74% of the variance of the dataset and reported no systematical bias
343 in calculated values (Figure 7). During the development, we removed from the dataset samples below 1000
344 vehicles day⁻¹, for the above-depicted rationale, and samples where not all the data were available.

345 The validation of the method (performed with the leave-one-out method due to the low number of samples)
346 gave also encouraging results, with an RPE value of 22% and a cross-validated coefficient of determination
347 (Q²) of 0.65, that indicate a good fit of the data. To have a better idea of the fit maintaining the original data
348 scale, the cross-validated standard error (SECV) was 0.52 mg m⁻². Converted as an error on emission factors,
349 it corresponds to 7 mg VKT⁻¹, a very encouraging result on maxima emission potentials, as we started from
350 mostly non-parametrized or city-averaged inventories.

351 Our model can be therefore used for predicting road dust loadings for multiple objectives: bottom-up emission
352 inventory, road maintenance for air quality management and air quality measures in general.

353

354 **Conclusions**

355 In this study, we explored the influence of road characteristics such as pavement macro-texture, traffic intensity
356 and distance to braking zones on real-world road dust loading that can be mobilized and resuspended (i.e.
357 emission factor) in two contrasting environments such as the cities of Turin and Barcelona. Our results showed
358 good inverse relationship between MF10 and i) macro-texture, ii) traffic intensity and iii) distance from
359 braking zone, although with a lower correlation. These results allowed building an empirical mode, able to
360 predict suspendable road dust and emission factors based on the aforementioned road data. This model can
361 significantly improve bottom-up emission inventory for spatial allocation of emissions and air quality
362 management to select those roads with higher emissions for mitigation measures. We also found a clear
363 seasonal effect of road dust emissions, with higher emission in summer likely due to the lower pavement
364 moisture, indicating that our model could be used only for predicting spatial variability of maxima emissions,
365 which then need to be scaled for meteorological effect, using methods accounting for weather and pavement
366 moisture.

368 **Acknowledgements**

369 The authors wish to acknowledge the Turin City Council, as all the samplings were made within the project
 370 “Suolo, Polveri, Particolato”, in the framework of the Torino Living Lab project of the city of Turin. We thank
 371 5T s.r.l., Dr. Marco Bono and Dr. Alberto Escrig for their help. Additional funding from AXA research fund
 372 is also acknowledged.

373

374 **References**

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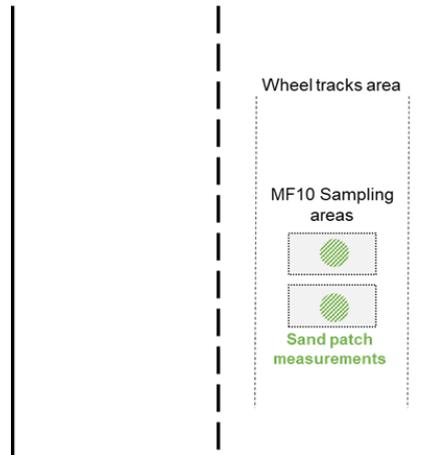
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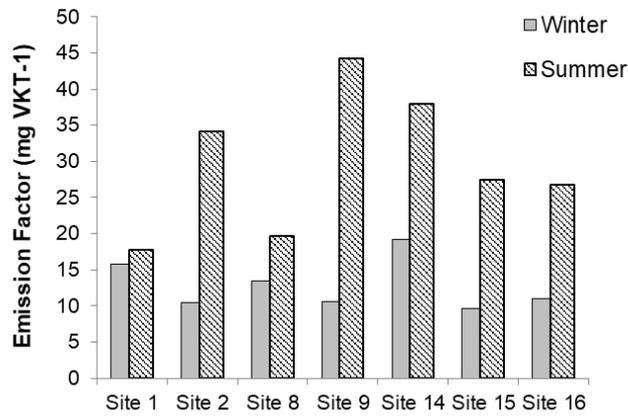
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Figure 1. Placement of the sampling area within the road.

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Figure 2. Comparison of summer and winter emission factors at selected Turin sites (where sampling was made at the same spot in both seasons).

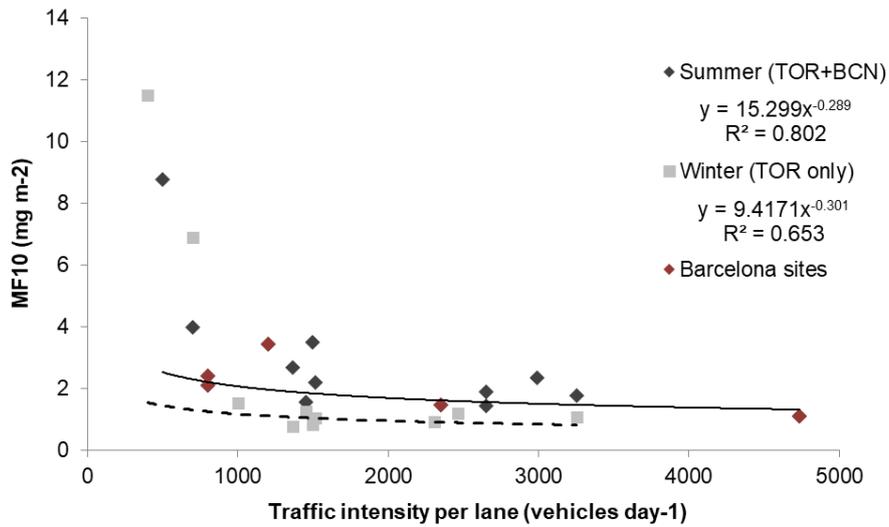
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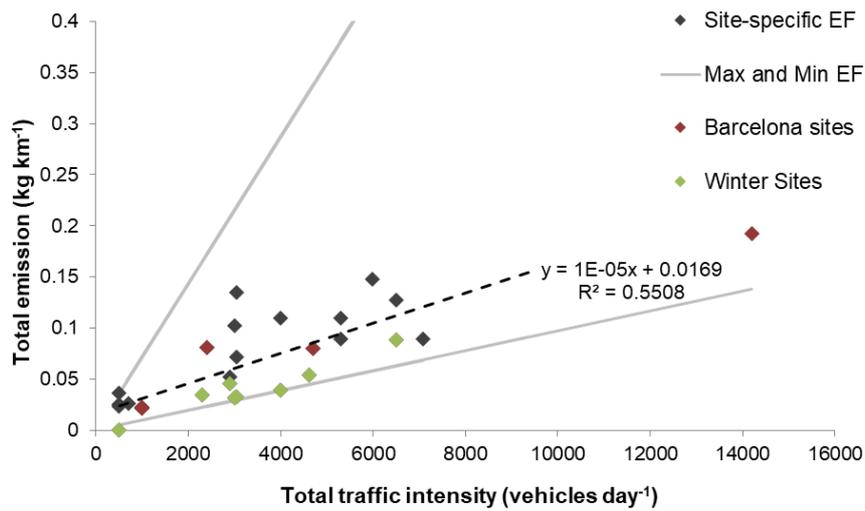
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535 Figure 3. Relationship between MF10 and traffic intensity per lane. All sites with available traffic intensity
 536 data were used. Red points belong to Barcelona sites.

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539 Figure 4. Site-specific relationship between total emission per km and traffic intensity (using data of both
 540 cities). Grey line represents the interval generated using the maximum and the minimum value of EFs. The
 541 site with the higher traffic intensity (in red) was excluded from the fit, although in line with other samples.

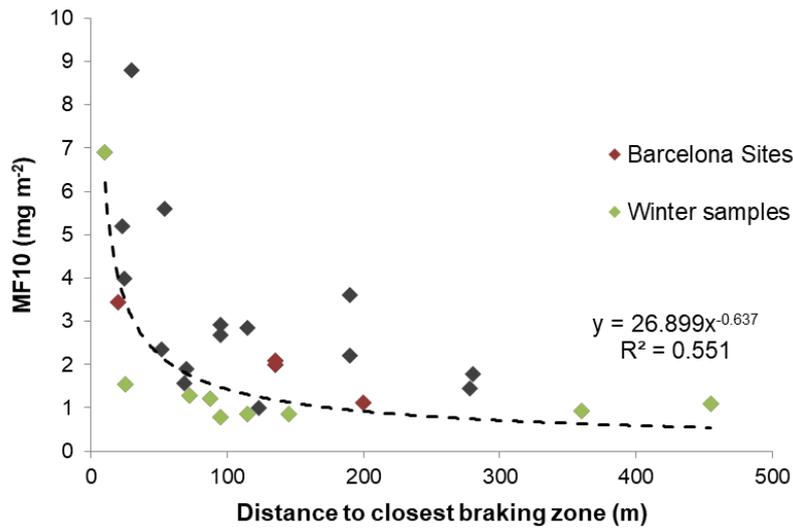
542 Samples in red are of Barcelona sites, while green samples belong to winter sampling.

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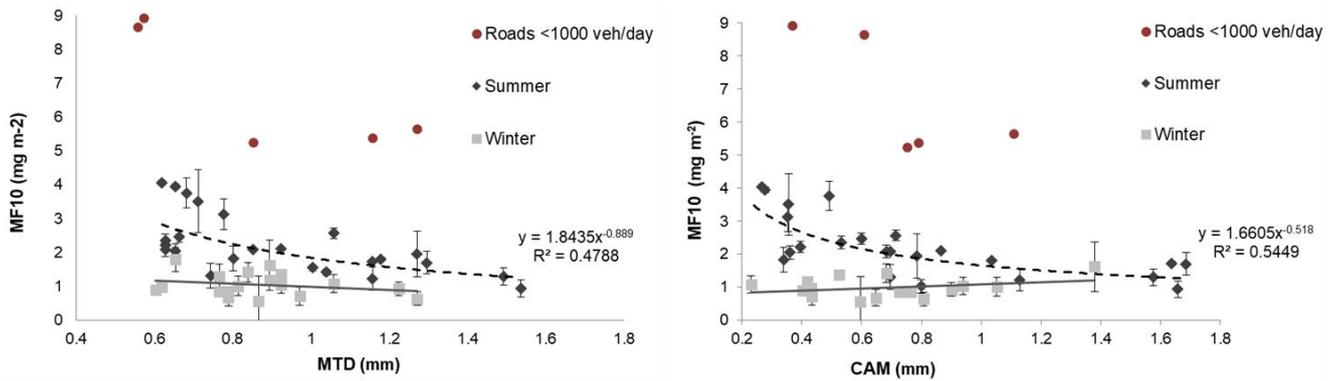
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548 Figure 5. Relationship between MF10 and the distance from the closest braking point: stop, traffic light or
 549 roundabout (using data of both cities). Samples in red are of Barcelona sites, while green samples belong to
 550 winter sampling.

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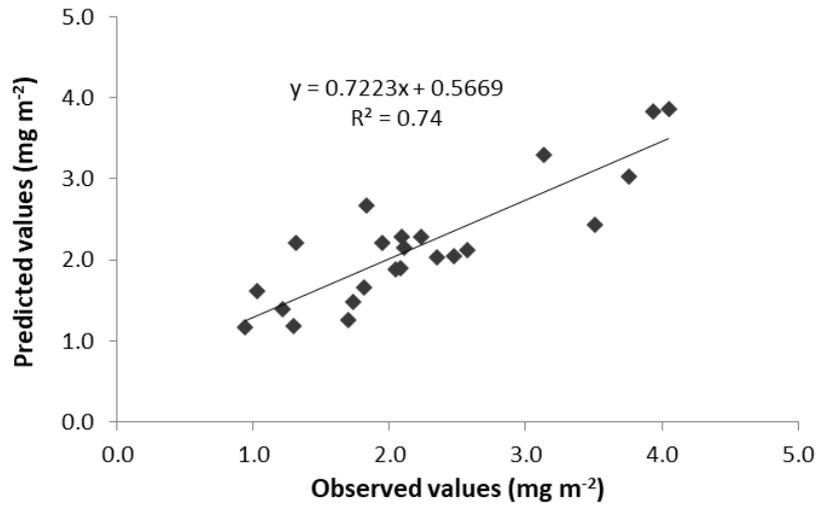
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553 Figure 6. Relationship between MF10 and MTD (left) and CAM (right) of aggregates. All samples have been
 554 used; red dots belong to sites where traffic intensity was below 1000 vehicles day⁻¹, and have been excluded
 555 from calculations.

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560 Figure 7. Observed versus predicted MF10 values, calculated using the linear regression presented in
561 equation 3. Sites where traffic intensity was below 1000 vehicles day⁻¹ have been excluded from the
562 calculations.

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Table 1. Site description, pavement MTDs, MF10 loadings and calculated emission factors for each sample (average of 2 filters).

* indicates traffic values estimated with mobility models.

Season	Site number (city)	Distance from closest braking zone [m]	N° vehicles per lane [veh day ⁻¹]*	Average speed (km/h)	MTD ± S.D (mm)	MF10 ± S.D. (mg m ⁻²)	EF ± S.D. (mg VKT ⁻¹)	Notes
Summer	1S (Turin)	68	1454	38.6	0.77±0.04	1.6±0.4	18±5	
	2S (Turin)	340	1500	25.5	0.68±0.05	3.5±0.5	34±6	
	3 (Turin)	103	500*	-	0.57±0.01	8.8±0.2	72±2	
	4 (Turin)	95	15367	36.9	-	2.9±1.0	36±13	
	5 (Turin)	52	2993	30.1	0.65±0.03	2.3±0.2	25±2	
	6 (Turin)	278	2651	-	1.23±0.12	1.5±0.3	17±4	
	7 (Turin)	280	2651	-	0.62±0.04	1.8±0.2	20±2	
	8S (Turin)	70	3255	-	1.17±0.1	1.9±0.1	21±1	
	9S (Turin)	210	1519	29.6	-	3.6±0.6	44±7	
	10 (Turin)	210	1519	-	0.63±0.02	2.2±0.2	24±2	
	11 (Turin)	123	3542	32.2	0.97±0.03	1.0±0.2	13±2	
	12 (Turin)	54	500*	-	1.27±0.04	5.6±0.5	50±6	
	13 (Turin)	23	500*	-	0.85±0.02	5.2±0.4	47±5	
	14S (Turin)	25	700*	-	0.64±0.03	4.0±0.1	38±1	
	15S (Turin)	155	1367	-	1.28±0.27	2.7±0.2	27±2	
	16S (Turin)	150	1000	-	-	2.2±0.2	27±3	

	17 (Turin)	-	-	-	-	5.5±0.8	68±10	University campus road
	18 (Turin)	-	-	-	-	12.3±1.6	152±20	Park road
	19 (Turin)	70	4700	-	-	42.7±5	528±62	Constructi on works nearby
	20 (Barcelona)	5	1200	-	0.73±0.08	3.4±0.4	34±5	
	21 (Barcelona)	80	2350	-	1.0±0.04	1.5±0.1	17±1	
	22 (Barcelona)	65	800	-	0.89±0.07	2.0±0.01	23±1	
	23 (Barcelona)	65	800	-	1.27±0.04	2.1±0.7	21±10	
	24 (Barcelona)	200	4733	-	1.52±0.08	1.1±0.2	14±3	
	25 (Barcelona)	95	-	-	0.40±0.05	1.1±0.2	13±3	
Winter	1W (Turin)	72	1454	38.6	0.91±0.07	1.3±0.1	16±2	
	2W (Turin)	340	1500	25.5	0.78±0.04	0.85±0	11±0	
	8W (Turin)	455	3255	-	0.88±0.07	1.1±0.6	14±9	
	9W (Turin)	260	1519	29.6	0.86±0.13	0.9±0.3	11±3	
	14W (Turin)	45	500*	-	0.71±0.07	1.6±0.4	19±4	
	15W (Turin)	155	1367	-	1.25±0.05	0.8±0.2	10±3	
	16W (Turin)	150	1000	-	1.02±0.10	0.9±0.3	11±3	
	26 (Turin)	130	500*	-	-	11.5±4.5	142±55	Constructi on works nearby
	27 (Turin)	93	500*	-	1.03±0.16	6.9±2.2	85±27	
	28 (Turin)	170	2464	14.7	0.83±0.03	1.2±0.3	15±4	
	29 (Turin)	1000	2311	23.1	0.61±0.04	0.9±0.1	12±1	

