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3 **Non-Exhaust Vehicle Emissions of Particulate Matter**
4 **and VOC from Road Traffic: A Review**
5

6
7 **Roy M. Harrison¹, James Allan², David Carruthers³,**
8 **Mathew R. Heal⁴, Alastair C. Lewis⁵, Ben Marner⁶,**
9 **Tim Murrells⁷ and Andrew Williams⁸**

10
11 **¹ Division of Environmental Health & Risk Management**
12 **School of Geography, Earth & Environmental Sciences**
13 **University of Birmingham, Edgbaston, Birmingham B15 2TT, UK**

14
15 **² Department of Earth and Environmental Sciences**
16 **University of Manchester, Williamson Building**
17 **Oxford Road, Manchester, M13 9PL, UK**

18
19 **³ Cambridge Environmental Research Consultants**
20 **3 King's Parade**
21 **Cambridge CB2 1SJ, UK**

22
23 **⁴ School of Chemistry, University of Edinburgh**
24 **Joseph Black Building, David Brewster Road, Edinburgh, EH9 3FJ, UK**

25
26 **⁵ Department of Chemistry, University of York**
27 **Heslington, York YO10 5DD, UK**

28
29 **⁶ Air Quality Consultants, 23 Coldharbour Road**
30 **Bristol BS6 7JT, UK**

31
32 **⁷ Ricardo Energy & Environment, Gemini Building, Fermi Avenue, Harwell,**
33 **Oxon, OX11 0QR, UK**

34
35 **⁸ Mechanical Engineering, Faculty of Science and Engineering**
36 **Thornton Science Park, University of Chester, Pool Lane, Ince, CH2 4NU, UK**
37
38

39 **ABSTRACT**

40 As exhaust emissions of particles and volatile organic compounds (VOC) from road vehicles have
41 progressively come under greater control, non-exhaust emissions have become an increasing
42 proportion of the total emissions, and in many countries now exceed exhaust emissions. Non-
43 exhaust particle emissions arise from abrasion of the brakes and tyres and wear of the road surface,
44 as well as from resuspension of road dusts. The national emissions, particle size distributions and
45 chemical composition of each of these sources is reviewed. Most estimates of airborne
46 concentrations derive from the use of chemical tracers of specific emissions; the tracers and
47 airborne concentrations estimated from their use are considered. Particle size distributions have
48 been measured both in the laboratory and in field studies, and generally show particles to be in both
49 the coarse (PM_{2.5-10}) and fine (PM_{2.5}) fractions, with a larger proportion in the former. The
50 introduction of battery electric vehicles is concluded to have only a small effect on overall road
51 traffic particle emissions. Approaches to numerical modelling of non-exhaust particles in the
52 atmosphere are reviewed. Abatement measures include engineering controls, especially for brake
53 wear, improved materials (e.g. for tyre wear) and road surface cleaning and dust suppressants for
54 resuspension. Emissions from solvents in screen wash and de-icers now dominate VOC emissions
55 from traffic in the UK, and exhibit a very different composition to exhaust VOC emissions. Likely
56 future trends in non-exhaust particle emissions are described.

57

58 **KEYWORDS:** non-exhaust emissions; road traffic; particulate matter, wear, resuspension,
59 emissions, volatile organic compounds

60

61

62 **1. INTRODUCTION**

63 Road traffic has long been viewed as a major contributor to urban air pollution. The advent of
64 three- way catalytic converters on gasoline vehicles has led to a huge reduction in exhaust
65 emissions of carbon monoxide, hydrocarbons and NO_x (Winkler et al., 2018). Reductions in
66 emissions from diesel exhausts came much later, and these remain a major source of particulate
67 matter and NO_x pollution in cities globally (Harrison et al., 2021). Diesel Particle Filters are a
68 highly efficient means of controlling particulate matter emissions from diesels, and the latest
69 particle number emissions standards in Europe require the use of Gasoline Particle Filters.
70 Consequently, particulate matter emissions from traffic exhaust in developed countries, where new
71 vehicles have to adopt these technologies, have declined rapidly, and as a consequence, non-exhaust
72 emissions from road vehicles now widely exceed exhaust emissions (AQEG, 2019). According to
73 estimates reported by OECD (2020), the total amount of non-exhaust particulate matter (PM_{2.5})
74 emitted by passenger vehicles worldwide is likely to rise by 53% by 2030 from a 2017 baseline,
75 mainly due to increased vehicle mileage.

76
77 Non-exhaust emissions of particles arise mainly from four separate sources. Three of these arise
78 from abrasion: brake wear, tyre wear and road surface wear. The fourth, particle resuspension, arises
79 from road dust particles being suspended into the atmosphere by passing traffic. There are also
80 emissions of VOC from use of screenwash, which are rarely considered. All of these emission
81 types are generated by battery-electric vehicles as well as internal combustion engine vehicles, and
82 although their magnitude may change, road traffic will continue to be a source of particulate matter
83 and VOC emissions, even for a fully electric vehicle fleet.

84
85 Measurements of non-exhaust emissions come from both controlled laboratory studies and field
86 experiments. Brake wear, tyre wear and road surface wear can all be evaluated in the laboratory.
87 Brake wear can be determined under highly controlled conditions using a brake dynamometer, and

88 standardised methods allowing comparative studies of materials are under development. Methods
89 for evaluating tyre and road surface wear are more difficult to standardise and there are a range of
90 test methods, with relatively few data available. Such studies are, however, important as they are
91 the basis for deriving the emissions factors which form the building blocks for emissions
92 inventories, and are essential for numerical modelling of airborne concentrations. Resuspension is
93 more difficult to estimate. It is often not included in emissions inventories, and most estimates are
94 based upon algorithms using vehicle mass and road surface dust loading as input data. Non-exhaust
95 particles can also be measured in the atmosphere by use of chemical tracers, and by using receptor
96 modelling methods such as Positive Matrix Factorization. This is challenging and relatively few
97 data exist, although these have increased recently.

98

99 Past reviews of this topic, or specific aspects of it, have previously been published by Thorpe and
100 Harrison (2008), Grigoratos and Martini (2015), Amato et al. (2014) and AQEG (2019). It is
101 however, a topical and rapidly moving field, and hence we feel it valuable to provide a brief review
102 of the latest state of knowledge.

103

104 **2. INVENTORIES OF NON-EXHAUST EMISSIONS**

105 Emission inventories for many countries include estimates of emissions of PM₁₀ and PM_{2.5} from
106 tyre wear, brake wear and road abrasion and, in some cases, for road dust resuspension. The
107 methods used are mainly suited for estimating emissions on a national or regional scale as they
108 simplify the effect of vehicle size, type and technology, driving style and road condition. This is
109 normally done out of necessity when national statistics on vehicle population and mileage and travel
110 on different types of roads at different average speeds are used to estimate emissions on a large
111 scale. These approaches are less well suited to estimating emissions on a particular road section at a
112 particular time where emissions are expected to be highly variable.

113

114 In Europe, countries report emission inventories annually to conform with guidelines set out by the
115 European Environment Agency. This is to ensure consistency and comparability in inventories
116 reported by EU Member States. The EMEP/EEA Emissions Inventory Guidebook (EMEP/EEA,
117 2019) provides methods and emission factors for estimating emissions from tyre wear, brake wear
118 and road abrasion for different vehicle types. Emission factors are in mg/km and correction factors
119 are provided to account for different average speeds for mileages done on urban, rural and highway
120 roads and different vehicle loads in the case of heavy duty vehicles. Emissions from resuspension
121 of previously deposited material are not required to be reported and no method is provided in the
122 Guidebook. This is partly because this would be considered a double-count of an emission for
123 accounting purposes, though it is recognised that this source does have to be considered when
124 considering the full impact of traffic on ambient concentrations of PM in urban environments.

125

126 The emission factors provided in the EEA Guidebook are mostly based on analysis of data from
127 literature sources of nearly 20 years ago on wear rates for tyre and brake material combined with
128 estimates of the quantities that become airborne in different particle size ranges, with some data
129 from direct measurement using a simulated wheel or brake operation in the laboratory. The
130 factors may not well represent emissions from modern vehicle technologies and tyre and brake
131 systems and materials. Countries are permitted to use country-specific factors if these can be
132 justified and take account of national circumstances. For example, the Netherlands and Sweden
133 consider a smaller proportion of tyre wear and brake wear PM emitted in the PM_{2.5} range relative to
134 PM₁₀ compared with the values given in the Guidebook. Sweden and Finland apply a larger
135 emission factor for PM₁₀ from road abrasion to account for the effect of studded tyres relative to
136 non-studded tyres and use a much smaller PM_{2.5}/PM₁₀ ratio for this source to reflect the high
137 proportion of these emissions in the coarse particle range (see for example, Finland's 2020
138 Informative Inventory Report for Transport at <https://www.environment.fi/en->

139 US/Maps_and_statistics/Air_pollutant_emissions/Finnish_air_pollutant_inventory_to_the_CLRTA
140 P). Further details on emission factors used in Europe are given in AQEG (2019).

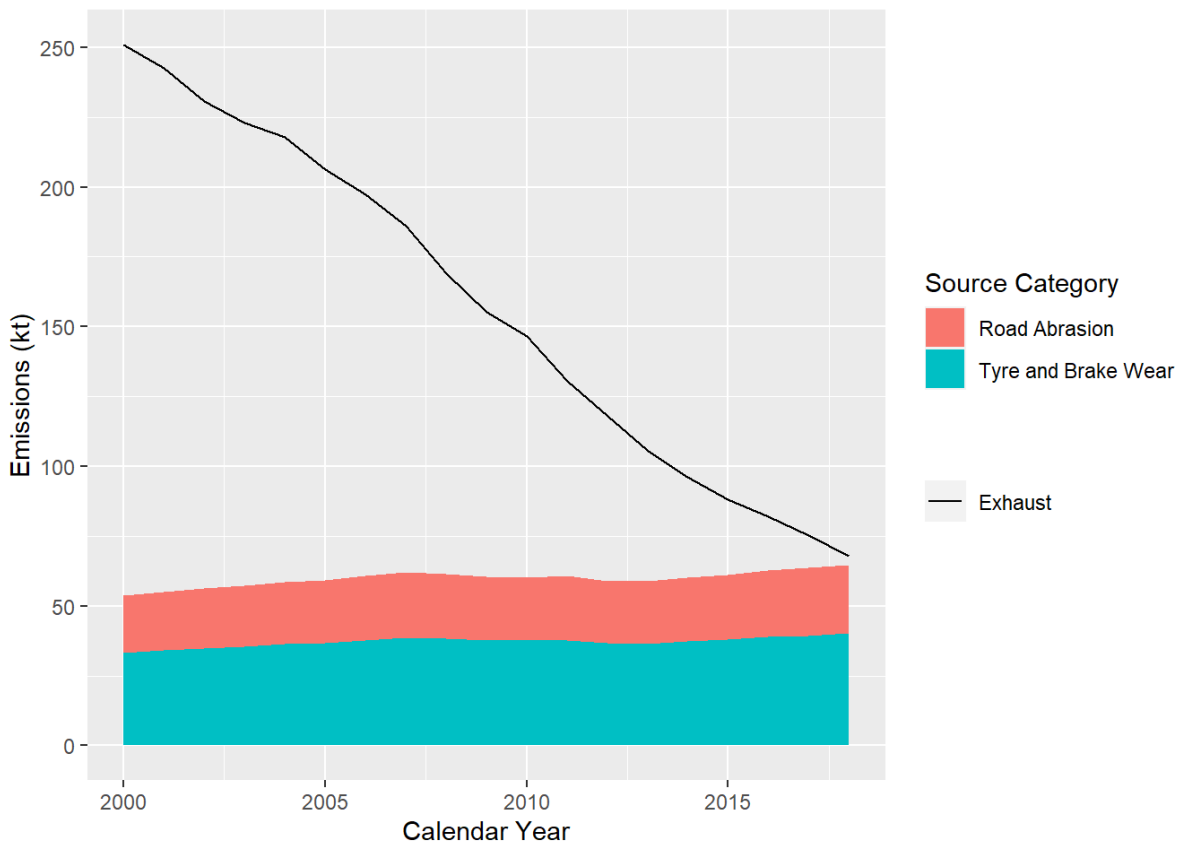
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142 Without any regulation on non-exhaust emissions or understanding of how modern vehicle
143 technologies have changed emissions, the emission factors used in Europe remain constant over
144 time. As a consequence, all inventories show emissions from these sources increasing over time as
145 the numbers of vehicles and kilometres travelled have increased. In contrast, exhaust emissions in
146 most countries have been decreasing as newer vehicles meeting tighter emission standards enter the
147 fleet, particularly in recent years as increasing numbers of diesel vehicles are fitted with particulate
148 filters.

149

150 Figure 1 shows the trend in exhaust and non-exhaust emissions of $PM_{2.5}$ for all EU28 countries
151 combined from 2000-2018 taken from inventories officially submitted by the Parties to the UNECE
152 Convention on Long-range Transboundary Air Pollution (EEA, 2021). Inventory data submitted
153 by countries combine emissions from tyre and brake wear in a single category, but road abrasion
154 emissions are reported separately. Whilst there are some differences between individual countries,
155 most are showing the same general trends as Figure 1. For most countries, each of the three non-
156 exhaust sources contribute roughly similar amounts and there is no dominant source, though
157 countries such as Sweden and Finland have a much greater contribution of road abrasion to the
158 PM_{10} inventory.

159



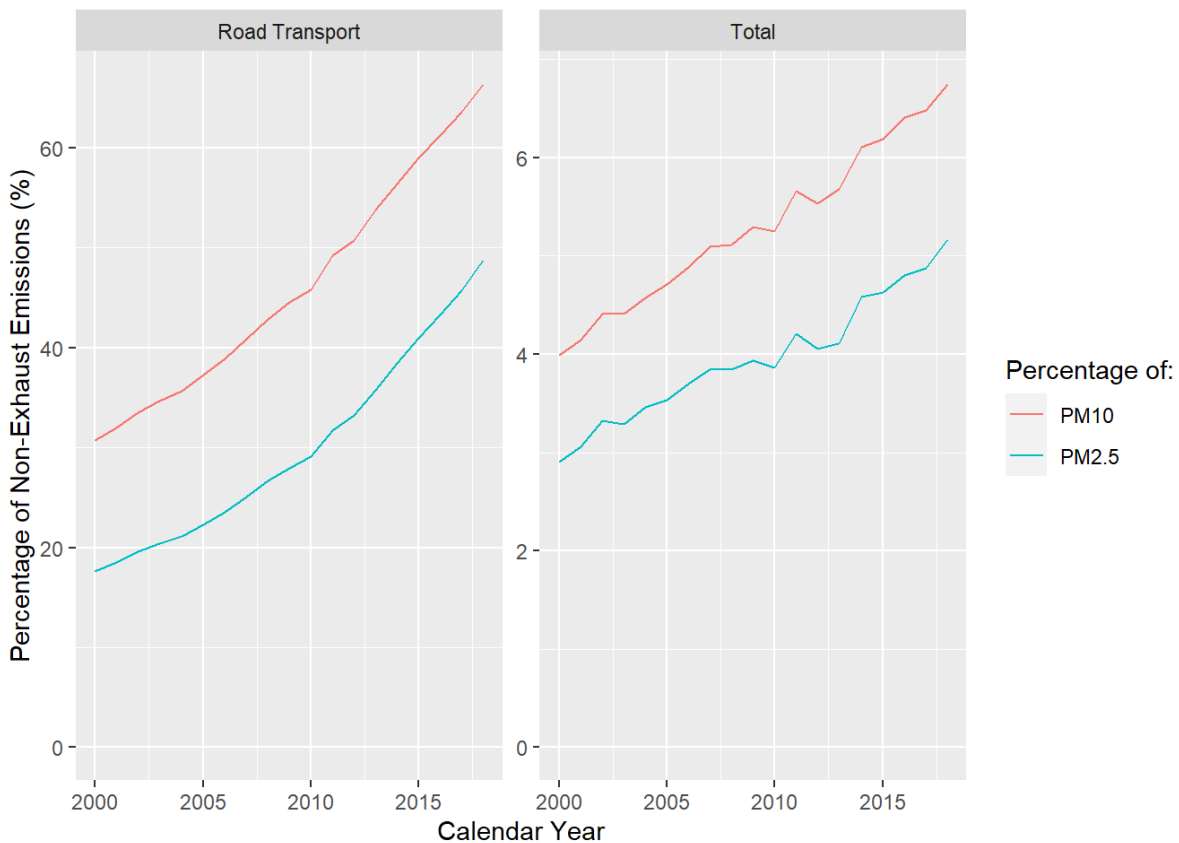
160

161 **Figure 1:** Total exhaust and non-exhaust road transport emissions of PM_{2.5} from EU28 countries
 162 from emission inventories reported by Parties under the Convention on Long-range Transboundary
 163 Air Pollution in 2020, covering years up to 2018 (EEA, 2021).

164

165 Figure 2 shows non-exhaust emissions of PM_{2.5} and PM₁₀ for EU28 countries combined as a
 166 percentage share of total road transport and of all national primary emissions. These trends imply
 167 that the share of road transport emissions from non-exhaust sources has been increasing with time,
 168 rising from 18% in 2000 to 49% in 2018 in the case of PM_{2.5} and 31% to 66% in the case of PM₁₀.
 169 As a share of all primary emissions in the EU28, non-exhaust emissions increase from 2.9% in 2000
 170 to 5.2% in 2018 in the case of PM_{2.5} and 4.0% to 6.7% in the case of PM₁₀.

171



172

173 **Figure 2:** Percentage share of total non-exhaust emissions of PM_{2.5} and PM₁₀ for EU28 countries
 174 combined, as a percentage share of total road transport and of all national primary emissions (shown
 175 in the figure marked ‘Total’). Based on data from EEA (2021).
 176

177 However, the range in the contribution made by non-exhaust emissions to total road transport
 178 emissions between countries is very wide, according to the official inventories of EU28 Member
 179 States (EEA (2021), these include the UK for the purpose of these comparisons). For PM_{2.5}, the
 180 range in 2018 varies from 11% to 87% and may reflect the different transport activities in each
 181 country as well as country-specific circumstances, such as the use of studded tyres in Sweden or it
 182 may reflect the use of different emission factors. The range in the PM_{2.5} share in emissions is
 183 smaller for the largest European countries (UK, France, Germany, Spain, Italy), from 42% to 67%
 184 of all road transport emissions in 2018.

185

186 The National Emissions Inventory for the U.S. includes emissions of PM₁₀ and PM_{2.5} from tyre and
 187 brake wear and resuspension. Emissions from tyre and brake wear are included with exhaust

188 emissions in the ‘On-road – Mobile’ category and cannot be viewed separately; emissions from
189 resuspension are reported separately under ‘Paved road dust’.

190

191 Tyre and brake wear emissions are estimated by the USEPA MOVES tool (USEPA, 2020a). Tyre
192 wear emissions in MOVES are derived from wear rates correlated with average vehicle speed,
193 similar to the approach used in Europe. It assumes that 8% of tyre wear is emitted as PM₁₀, 1.5% as
194 PM_{2.5}. This leads to an average PM₁₀ factor for passenger cars of 5 mg/km which is comparable to
195 the values used in Europe, but the average value for trucks is lower than European values at less
196 than 20 mg/km. This may be due to different assumptions about vehicle sizes and average speed
197 and how tyre wear emissions are correlated. The US approach implies a much smaller tyre wear
198 PM_{2.5}/PM₁₀ ratio of 0.15 on average (USEPA, 2020a), compared with 0.7 in Europe (EMEP/EEA,
199 2019).

200

201 MOVES uses g/hour emission rates for brake wear based on brake dynamometer and wind tunnel
202 measurements at different deceleration rates for US-specific brake materials (USEPA, 2020a).

203 These are combined with estimates of the amount of braking done at different deceleration levels
204 for different real-world vehicle operating modes defined in terms of vehicle specific power, a more
205 detailed approach than the Guidebook approach used in Europe. The average brake wear emission
206 factor for PM₁₀ calculated in MOVES for passenger cars in the US inventory is quoted as 14 mg/km
207 which is higher than the UK average of 7mg/km derived from the UK’s national inventory (NAEI,
208 2020) using factors from EMEP/EEA (2019), but MOVES implies a much smaller brake wear
209 PM_{2.5}/PM₁₀ ratio of 0.12 on average (USEPA, 2020a), compared with 0.4 in Europe (EMEP/EEA,
210 2019).

211

212 Emissions from resuspension in the U.S. inventory, categorised under the term ‘paved road dust’
213 are calculated in a method described in the USEPA’s AP-42 compilation manual (USEPA, 2011).

214 A near-linear empirical equation is used that relates an emission factor in g/km to vehicle mass and
215 silt loading of the road surface. A correction is made for the number of precipitation days in the
216 model period. Road surface and environmental conditions are highly variable and the factors from
217 this approach are highly uncertain and not necessarily universal as it applies only to the range of
218 variables from which the equation was derived. The applicability to other conditions of vehicle
219 speed, mass and road conditions may be limited. AP-42 provides an alternative equation for
220 unpaved roads.

221

222 Comparing non-exhaust emissions from on-line national inventory data sources for the U.S. and
223 Europe is difficult on a like-for-like basis because the scope of coverage and accessibility of data
224 are different. Europe shows tyre and brake wear emissions (combined) and road abrasion emissions
225 separately, but excludes resuspension; the U.S. includes resuspension (paved road dust), but
226 combines tyre and brake wear with exhaust emissions. A report by Panko et al. (2013) states that
227 according to the USEPA, tyre wear contributed 15% of total vehicle emissions of PM₁₀ in 2008
228 which compares with a value of 17% in the UK for the same year (NAEI, 2020). On-line inventory
229 data are available for the State of California where emissions for all sources can be seen with tyre
230 and brake wear and resuspension (paved road dust) viewed separately. Combining data extracted
231 from the CARB mobile emissions data source EMFAC (CARB, 2017) and the emissions tool
232 CEPAM (CARB, 2016) for all other sources in 2018, the percentage contributions of tyre and brake
233 wear and resuspension relative to all road transport emissions and as a percentage of all California
234 emissions can be derived.

235

236 Table 1 compares these contributions for the UK, Europe, California and the U.S. as far as possible
237 on a like-for-like basis. A comparison between the UK, Europe and California excludes
238 resuspension as this is not covered in European inventories. Only emissions for resuspension can
239 be shown for the U.S. The table shows comparisons in contributions as a percentage of all road

240 transport emissions in the country/region and as a percentage of all primary PM emissions for the
 241 year 2018, apart from the U.S. where the most recent data is for 2017.

242
 243 **Table 1:** Contribution of non-exhaust sources to road transport and all primary sources of PM
 244 emissions in Europe and the U.S. These are according to current estimates from emission
 245 inventories for the respective regions, with comparisons done on a like-for-like basis, as far as is
 246 possible. Data are derived for the UK from NAEI (2020), for EU28 from EEA (2021), for
 247 California from CARB (2016) and CARB (2017) and for the USA from USEPA (2020a).
 248

				PM_{2.5}	PM₁₀
UK	2018	Tyre, brake, road abrasion	as % of all road transport emissions (exc resuspension)	67%	79%
		Tyre, brake, road abrasion	as % of all primary source emissions (exc resuspension)	8%	9%
EU28	2018	Tyre, brake, road abrasion	as % of all road transport emissions (exc resuspension)	49%	66%
		Tyre, brake, road abrasion	as % of all primary source emissions (exc resuspension)	5%	7%
California	2018	Tyre & brake	as % of all road transport emissions (exc resuspension)	71%	85%
		Tyre & brake	as % of all primary source emissions (exc resuspension)	7%	5%

249

				PM_{2.5}	PM₁₀
USA	2017	Resuspension	as % of all road transport emissions	65%	79%
		Resuspension	as % of all primary source emissions	4%	5%
California	2018	Resuspension	as % of all road transport emissions	46%	72%
		Resuspension	as % of all primary source emissions	7%	11%

250
 251 Emissions as a fraction of all primary emissions in the country/region will reflect the quantity of
 252 emissions from non-transport sources which can be quite variable. Even so, it is clear to see that
 253 resuspension alone makes a large contribution to area-wide emissions from road transport of PM_{2.5}
 254 and PM₁₀ in the U.S., according to the inventories. In Europe, where emissions from resuspension
 255 are excluded from inventories, it is clear how even emissions from the other non-exhaust sources
 256 make large contributions to total road transport emissions of PM, often more than 50% in 2018, and
 257 this is consistent with the situation in California where the contributions from tyre and brake wear
 258 may be even greater. The inventory for California suggests that when all non-exhaust sources are

259 accounted for, including resuspension, they contribute to 96% of all emissions of PM₁₀ from road
260 traffic and 15% of primary PM₁₀ emissions from all sources in 2018 (CARB, 2016, 2017).

261

262 There are a few reported estimates of non-exhaust emissions from road transport in China. Zhang
263 et al. (2020) recently conducted a comprehensive study combining tunnel measurements, laboratory
264 dynamometer and resuspension experiments and chemical mass balance modelling to estimate road
265 dust resuspension, tyre wear and brake wear emissions in four Chinese megacities. The
266 experiments yielded emission factors for PM_{2.5} not dissimilar to the factors used in Europe for tyre
267 and brake wear, but the authors acknowledged their factors could be underestimates, not typical of
268 most traffic in China because of the smooth traffic flow and low amount of braking in the tunnels.
269 The authors also concluded a total inventory for non-exhaust emissions of PM_{2.5} in China of 11.4
270 ktonnes based on measurements done in 2017/18, of which 8.1 ktonnes were from resuspension
271 (Zhang et al, 2020). Although it is not clear whether this value only applied to light duty traffic, the
272 value seems low when one considers the tyre and brake wear and road abrasion emissions alone in
273 the UK were 8.7 ktonnes in 2019 (NAEI, 2020), emissions from resuspension on paved roads were
274 209 ktonnes in the U.S. in 2017 (USEPA, 2017) as well as the fact this would be small relative to
275 estimates of total road transport emissions of PM₁₀ in China ranging from 40 – 400 ktonnes in 2008,
276 depending on inventory source (Saikawa et al., 2017).

277

278 An air pollutant emissions inventory is available for road transport in 2013 in Australia (Smit,
279 2014), developed using the same method as used in Europe, from the EMEP/EEA Emissions
280 Inventory Guidebook. The inventory suggests non-exhaust emissions were responsible for 26% of
281 all road transport emissions of PM_{2.5}, similar to the contributions in Europe for 2013 (EEA, 2021).

282

283 The OECD has published a highly detailed review of non-exhaust emissions of PM which considers
284 the causes, consequences and policy responses to tackle these emissions globally (OECD, 2020).

285 The report considered the limitations of emission inventories, including the inclusion or otherwise
286 of resuspension, and reviews inventories for different regions of the world, mainly Europe and the
287 U.S., but also in parts of Latin America. Inventories for Mexico, Bogota in Colombia, and Santiago
288 in Chile, all highlight the importance of road dust resuspension with contributions to total primary
289 emissions from all sources ranging from 16% to 54% (PM₁₀) and 9% to 56% (PM_{2.5}) in these
290 regions. The contribution of tyre and brake wear emissions to total emissions in Bogota was much
291 less than that, at 1.4% for PM₁₀ and 0.8% for PM_{2.5}.

292

293 In a review of tyre wear and its impact on the environment, Kole et al. (2017) estimated the global
294 emission of tyre wear to be 5.9 million tonnes per year. On the assumption that 3-7% of this
295 becomes airborne PM_{2.5} emissions (Kole et al., 2017; Grigoratos and Martini; 2014), a global
296 emission rate of 175-410 ktonnes PM_{2.5} from tyre wear is implied.

297

298 The OECD conclude that globally in 2014 at least 50% of total traffic emissions of PM₁₀ and 5% of
299 primary PM₁₀ from all sources were non-exhaust emissions, even when resuspension is excluded.
300 Corresponding figures for PM_{2.5} are 34% and 4%, respectively. When resuspension is taken into
301 account, non-exhaust emissions represent 95% of all road traffic emissions and 15% of all source
302 emissions, as stated earlier for California.

303

304 Non-exhaust sources do not only contribute to airborne PM₁₀ and PM_{2.5}, but also various metals and
305 polyaromatic hydrocarbons (AQEG, 2019). Emissions are generally related to the composition of
306 materials used in tyres and brake linings. Inventories include emissions of several different metals
307 and PAH species based on factors from fairly old literature sources, though some countries use rates
308 based on more recent composition studies. Some countries take account of the recent prohibition of
309 “PAH-rich” extender oils used in tyre production leading to a reduction in PAH emissions from tyre
310 wear (AQEG, 2019).

311 There are large uncertainties in emission inventories for non-exhaust sources. This is in part due to
312 the difficulty in measuring emissions, but also because of the high variability in emissions and the
313 factors which influence them. These are referred to in the studies mentioned above and in
314 Guidebooks and reports describing inventory methodologies. Emissions depend on driving style,
315 speed and the amount of acceleration and deceleration and cornering, weight and load of the
316 vehicle, the design of the brake system, type and composition of tyre and brake materials, the
317 condition and roughness of the road surface and ambient temperature, precipitation and humidity.
318 Inventories necessarily approximate and average out their effects or parameterise them in fairly
319 simple ways. Emissions from brake wear are not continuous but are inevitably greatest where
320 braking events are most significant. Methods have been developed to map brake wear emissions at
321 high spatial resolution where there are areas of high braking intensity according to changes in
322 vehicle specific power based on real-world vehicle measurements data associated with different
323 road situations (FAT, 2019). However, these methods have not yet been applied to inventories.

324

325 **3. PHYSICAL PROPERTIES OF NON-EXHAUST EMISSIONS**

326 **3.1 Brake Wear**

327 Brake wear particles arise from wear of both the brake lining (pad) and the rotor (disc) or drum.
328 There are three major types of brake lining, referred to as non-asbestos organic (NAO), low metallic
329 (LM) and semi-metallic (SM) (Kukutschova and Filip, 2018), each with a different composition and
330 favoured in different markets. Ceramic linings are also used on some of the more expensive
331 vehicles (Ostermeyer and Muller, 2008). The lining is made up from four main components:
332 binders, fibres, fillers and friction modifiers, and according to Roubicek et al. (2008), more than
333 3000 materials are used across brands. This is reflected in a very wide range of chemical
334 composition exemplified by the study of Hulskotte et al. (2014). The compositional differences
335 affect the rate of particle emissions (Kim et al., 2020), and most probably the physical properties of
336 emissions, but this has not been systematically studied. At normal brake operating temperatures

337 (<150°C), brake wear particles are formed by abrasion due to friction between the pad and rotor or
338 drum. They are irregular in shape with strong angular features, and contain multiple elements in the
339 same particle (Wahlstrom et al., 2010a). Exceptionally, at higher temperatures, ultrafine particles
340 are formed, and appear broadly similar in shape to the larger particles (Wahlstrom et al.,
341 2010a). Many studies have been conducted under controlled conditions, giving a range of results for
342 median diameters, which may arise from different designs of measurement system, or different
343 brake materials. In tests of various brake linings including both NAO and SM types using a brake
344 dynamometer, Garg et al. (2000) reported highly varied particle mass size spectra, which were
345 attributed by Sanders et al. (2003) in part to particle losses in the sampling system. Garg et al.
346 (2000) reported that between 7 – 48% of the particles became airborne, and at a brake temperature
347 of 100°C, of the particles sampled by a MOUDI impactor, 80%, 68% and 26% of mass was below
348 10 µm, 2.5 µm and 0.1 µm respectively, with a mass median diameter of 2.49 +/- 3.47 µm. Sanders
349 et al. (2003) measured from NAO, LM and SM types on a brake dynamometer and found particle
350 mass size spectra with medians at around 5 -6 µm and modes at 3-4 µm, despite the use of different
351 braking scenarios. Iijima et al. (2008) studied NAO linings and reported a mass concentration mode
352 at 3 - 6 µm. In tests of NAO linings, Hagino et al. (2016) found 2 – 21% of brake wear particles
353 became airborne, and particle mass was predominantly in the 0.8 – 2.2 µm size range. Various
354 brake pad and disc materials were tested by Wahlstrom et al. (2009; 2010b) in a laboratory pin-on-
355 disc rig, who found a mode in the mass distribution around 3 µm.

356

357 Brake wear size distributions have also been estimated from field measurements. In a receptor
358 modelling study, Wahlin et al. (2006) apportioned a range of elements to the brakes source:
359 chromium (Cr), iron (Fe), copper (Cu), zinc (Zn), zirconium (Zr), molybdenum (Mo), tin (Sn),
360 antimony (Sb), barium (Ba) and lead (Pb), and reported that particles created by brake abrasion had
361 mass median aerodynamic diameters around 2.8 µm. In a Positive Matrix Factorization (PMF)
362 study of wide range particle size spectra at a roadside site, Harrison et al. (2011) identified a source

363 of airborne particles which had a mode at 3 μm diameter and was associated with traffic-influenced
364 wind directions and heavy duty vehicle numbers which they concluded was caused by brake wear.
365 Similarly, analysing metals data from the same site (Marylebone Road, London), Harrison et al.
366 (2012) report a mass modal diameter of 3 μm for Ba, used as a tracer of brake wear. Similar modes
367 were seen for other elements such as Fe, Cu and Sb, associated with brake wear.

368

369 Sanders et al. (2003) in dynamometer and test track studies reported considerable difference in total
370 brake wear rates with low metallic (LM) pads emitting 3-4 times the material of semi-metallic (SM)
371 and non-asbestos organic (NAO) linings, but a consistent presence of Fe, Cu and Ba and a mass
372 median diameter of ca. 6 μm . Working in road tunnels in the United States, Lough et al. (2005)
373 measured mass modes at 3-6 μm diameter for brake-associated metals (Cu, Ba, Fe, Sb). According
374 to Gietl et al. (2010), the dominant mass mode of iron, copper, barium and antimony found at
375 Marylebone Road, London was between 1.2 and 7.2 μm aerodynamic diameter, and peaked at
376 around 3 μm . In analysis of antimony mass size spectra, Iijima et al. (2009) identified a bimodal
377 structure and attributed a mode at 3.6-5.2 μm as being due to brake dust. In the vast majority of
378 these studies, the modal diameter of brake wear particles is close to 2.5 μm , the diameter
379 conventionally taken in regulations as the boundary between fine and coarse particles. The
380 consequence is that when apportioning particulate matter mass into fine ($\text{PM}_{2.5}$) and coarse ($\text{PM}_{2.5-10}$)
381 fractions, significant quantities of brake wear particulate matter is recorded in both fractions.
382 Minor differences in experimental conditions can therefore result in substantial differences in the
383 $\text{PM}_{2.5}:\text{PM}_{2.5-10}$ ratio as seen in studies above (Garg et al., 2000; Iijima et al., 2008). Nosko and
384 Olofsson (2017) report brake wear particles to have an effective density ($0.75 \pm 0.2 \text{ g cm}^{-3}$), far
385 lower than the bulk density of the constituent material, and concluded that the particles are porous.

386

387 There has been much recent interest in emissions of nanoparticles from brake wear. Since these are
388 very small (defined as $<100\text{nm}$ diameter), when present they typically account for a very large

389 number of the particles, but very little mass. In an early study of the wear of brake components,
390 Roubicek et al. (2008) identified the emission of nanoparticles, using emission of CO₂ from the
391 brake pad as a measure of wear. Using a laboratory pin-on-disc apparatus, Wahlstrom et al. (2009;
392 2010b) found modes in the number distribution of particles emitted from various brake pad and disc
393 materials of around 350 nm and 550 nm, with a tail extending to much smaller sizes. As the
394 nanoparticle fraction is likely to arise from vaporisation-condensation of metals, rather than
395 abrasion, it is unsurprising that subsequent research has shown that emissions increase sharply
396 above a critical temperature (160-190°C) (Mathissen et al., 2018; Gonet and Maher, 2019). It is
397 reported that this temperature is not normally achieved during on-road vehicle usage (Farwick zum
398 Hagen et al., 2019a,b). In a pin-on-disc study, such particles could be as small as 1.3-10 nm
399 diameter (Nosko et al., 2017). Consistent with this, Mathissen et al. (2011) identified a unimodal
400 particle number size distribution near the disc brake, with a maximum at 11 nm.

401

402 In measurements within a road tunnel, Lough et al. (2005) observed a sub-micron mode in
403 atmospheric measurements of Cu, Fe, Pb and Ca, while Gietl et al. (2010) reported a secondary
404 mass mode of Cu and Sb in roadside tests at a diameter of 0.2 – 0.4 µm. These appear likely to be
405 associated with brake wear, although other sources, such as combusted lubricating oil may also
406 contribute for some elements such as Ca and Zn which are present in oil additives. Iijima et al.
407 (2008) reported a number concentration mode at 0.8 µm in brake dust spectra, but in a subsequent
408 paper, Iijima et al. (2009) suggest that a mode in the Sb size spectrum at 0.5 – 0.7 µm may have
409 been due to fly ash from waste incineration.

410

411 **3.2 Tyre Wear**

412 There are limited data on tyre wear particle emissions when compared with brake wear. Abrasion
413 occurs due to shear and friction forces arising from the contact between the vehicle wheel and the
414 road surface, and there is observational evidence of particle emissions containing fragments of road

415 surface material embedded in tyre wear particles, referred to as Tyre Road Wear Particles (TRWP).
416 The rate of tyre wear is affected by road surface roughness by factors of up to two to three
417 (Kennedy et al., 2002), and increased emissions may occur when the road surface is unpaved, or
418 especially when studded tyres are in use (Gustafsson et al., 2009). The latter are used to give grip in
419 icy conditions, and work has been carried out in Scandinavia (Johansson et al. 2012; Denby et al.,
420 2013) to predict the emissions from roads subject to damage from studded tyres and the addition of
421 grit to aid traction during winter. This will include both tyre and road surface wear.

422

423 Electron microscopy can reveal the composition and morphology of tyre dust particles. Studies by
424 Kreider et al. (2010) and Panko et al. (2013) have shown from on-road studies the association of
425 tyre rubber internally mixed with other constituents such as road surface wear, road dust or other
426 traffic-related sources within individual particles, known as Tyre Road Wear Particles (TRWP).
427 Aerosol Time-of-Flight Mass Spectrometry (ATOFMS) is an alternative form of single particle
428 analysis and has also shown an association of crustal elements with particles derived from tyre dust
429 (Dall'Osto et al., 2014). The particles are typically elongated with sausage-like shapes (Baensch-
430 Baltruschat et al., 2020; Klockner et al., 2021).

431

432 The majority of the mass of tyre wear particles is relatively large (Nielsen et al., 2003; Adachi and
433 Tainosho, 2004; Kreider et al., 2010) including elongated particles in excess of 100 μm in size, with
434 metal and mineral particles embedded within them. Kennedy et al. (2002) estimated that 80% of
435 tyre wear debris is deposited on the carriageway surface, with only 20% becoming airborne (as
436 TSP). According to ten Broeke et al. (2008), 5% of tyre wear occurs in the PM_{10} fraction.
437 Pischinger and Rauterberg-Wulff (1999) used elemental carbon and low volatility organic carbon to
438 distinguish between diesel exhaust and tyre debris in PM_{10} and $\text{PM}_{2.5}$ samples collected in a
439 highway tunnel, and report tyre wear particles only in the coarse fraction ($\text{PM}_{2.5-10}$).

440

441 Laboratory studies have examined the airborne particles arising from tyre wear. Dominant mass
442 modal diameters of tyre emission particles have been found at 2 μm (Sjodin et al., 2010) and at 2 –
443 5 μm (with an average size of 3.8 μm) by Raisenen et al. (2003). Kupiainen (2007) observed a mass
444 size spectrum with mode at 5 – 8 μm when operating a friction tyre on the rig used by Gustafsson et
445 al. (2009). Raisenen et al. (2003) found that the particle size distribution was not dependent on
446 asphalt or aggregate mineralogy. In tests on a laboratory rig and in on-road tests, Kreider et al.
447 (2010) found particle volume size spectra with modes at 50 μm to 100 μm , and bimodal particle
448 number concentrations with modes at 25 μm and 5 μm in the on-road data. They do not directly
449 report the fraction of particles below 10 μm or 2.5 μm , but it can be inferred from the volume size
450 spectra which show it to be very small. Park et al. (2018) working on a tyre wear simulator
451 observed a mass modal diameter of around 2 μm , which broadened under harsh friction conditions.
452 Kim and Lee (2018) found tyre wear modal diameters of 3 and 4 μm in unimodal distributions,
453 which is close to the mode in Zn concentrations attributed primarily to tyre wear measured on
454 Marylebone Road, London by Harrison et al. (2012).

455

456 Tyre wear is also able to generate nanoparticle emissions (<100 nm diameter). Mathissen et al.
457 (2011) measured particle number concentrations in the 6 – 562 nm size range near the contact point
458 on the road surface of a car travelling on a test track. This revealed a mode in the number size
459 distribution at 30 – 60 nm (mobility diameter) during acceleration, braking and extreme cornering.
460 During full lock braking, smoke was visible behind the tyre and the modal diameter increased to 60
461 – 70 nm. There was a general relationship observed between particle number concentration and
462 speed. An ultrafine (nanoparticle) mode in the tyre wear size spectrum has also been reported by
463 Kwak et al. (2014), Kim and Lee (2018), Foitzik et al. (2018) and Park et al. (2017). When
464 measuring ultrafine particle number emissions of studded and winter tyres on a laboratory rig, Dahl
465 et al. (2006), found that particle number emissions in the 15 – 700 nm size range increased with
466 vehicle speed, and that the smallest particles are volatile. This implies that the nanoparticle fraction

467 arises from a vapourisation/ condensation process involving oils used in the tyre, in contrast to the
468 larger particles which arise from abrasion (Kwak et al., 2014). This is consistent with a threshold
469 tyre tread temperature of 160°C for generation of this mode reported by Park et al. (2017), with
470 increased nanoparticle generation as the temperature increased.

471

472 **3.3 Road Surface Wear**

473 As explained above, road surface wear particles are normally emitted internally mixed in Tyre Road
474 Wear Particles (TRWP) and hence particle size spectra measured for tyre wear under field or
475 realistic laboratory conditions are likely to be a good representation for road surface wear.

476 However, there have been studies with a mobile load simulator and a road simulator. Gehrig et al.
477 (2010) operated a mobile load simulator and found road wear particles to have a main mode at 6 – 7
478 μm , and a smaller shoulder at ca. 12 μm . Operating a road simulator and studded tyres, Gustafsson
479 and Johansson (in Gustafsson, 2018) reported a mode at 5-6 μm , with particle mass orders of
480 magnitude greater than the mobile load simulator with conventional tyres. The main constituent of
481 road surfaces is mineral material (Gustafsson, 2018) which is unlikely to be differentiated from
482 resuspension on the basis of its chemical composition.

483

484 **3.4 Resuspension Particles**

485 A number of studies have sought sample resuspension by placing a sample inlet behind a tyre on a
486 moving vehicle. These have found a mode at 3 – 5 μm , regardless of whether a standard (summer)
487 or studded tyre was used (Hussein et al., 2008). Lee et al. (2013) report a number mode at 800 nm
488 with a concentration that did not vary with vehicle speed, while the mass mode at 2 – 3 μm saw a
489 slight increase in concentration with vehicle speed. A larger mass mode in the 6 – 10 μm diameter
490 range was found from sampling in a road tunnel by Lough et al. (2005) for the geological elements
491 magnesium (Mg) and calcium (Ca), most likely attributable to resuspension, or possibly road
492 surface wear. Harrison et al. (2011) applied Positive Matrix Factorization to wide range particle size

493 spectra measured on Marylebone Road, London, and identified a factor with a mode at diameter
494 $>10\ \mu\text{m}$ which occurred at traffic-influenced wind directions, and during the morning rush hour
495 when Heavy Duty Vehicle flow was greatest. Using Si as a tracer, Harrison et al. (2012) identified
496 the contribution of resuspension emissions on Marylebone Road, London, finding a mass modal
497 diameter of around $5\ \mu\text{m}$, coarser than the brake or tyre wear, extending to sizes beyond $10\ \mu\text{m}$.

498

499 **4. CHEMICAL COMPOSITION OF NEE**

500 **4.1 Brake Wear Particles**

501 Brake linings of frictional braking systems can be described as metallic, semi-metallic and non-
502 asbestos organic (NAO) according to the proportions of metallic and mineral and/or organic
503 material present (Chan and Stachowiak, 2004; Sanders et al., 2003). Across these classifications, the
504 chemical components in the brake linings typically derive from the following five categories of
505 function, listed here in decreasing order of contribution by mass (Grigoratos and Martini, 2015;
506 Thorpe and Harrison, 2008).

- 507 • Fillers (15-70%): cheaper materials to provide bulk and which include minerals such as barite
508 (BaSO_4), calcite (CaCO_3) and mica (silicates), and particles of rubber.
- 509 • Binders (20-40%): to maintain structural integrity of the brake linings under mechanical and
510 thermal stress and which typically comprise phenolic resins.
- 511 • Fibres (6-35%): to provide mechanical strength and which may be composed of various metals,
512 carbon, glass, Kevlar and, to a lesser extent, ceramic fibres.
- 513 • Friction modifiers or lubricants (5-29%): to stabilise the frictional properties and which are
514 typically composed of graphite, metal sulphides such as antimony trisulphide (Sb_2S_3), and
515 metal oxides.
- 516 • Abrasives (up to $\sim 10\%$): to increase friction and which may comprise a variety of substances
517 including aluminium oxide, iron oxides, silicon oxides (quartz), zirconium oxide and zirconium
518 silicate.

519 The broad range and quantities of chemical components listed above reflects the wide variety of
520 formulations used in brake linings in practice (Blau, 2001; Chan and Stachowiak, 2004), which is
521 also driven by commercial confidentiality. The plethora of materials used in brake liners means that
522 particles emitted from the braking process likewise contain a multitude of chemical components
523 (Grigoratos and Martini, 2015; Kukutschová et al., 2011). In addition, because of the wide variation
524 in resistance to mechanical disintegration and volatilisation across the chemical constituents, the
525 composition of brake-wear particles significantly differs from that of the original lining material
526 (Kukutschová et al., 2011; Österle et al., 2001). A substantial increase in ultrafine particle number
527 emissions when localised brake temperatures exceed $\sim 170^{\circ}\text{C}$, which is attributable to evaporation
528 and condensation of brake binder material (Perricone et al., 2018), may not contribute significantly
529 to mass of brake-wear emissions.

530

531 As the elemental content of brake pads is extremely variable (Hulskotte et al., 2014), almost the
532 complete suite of metallic elements (plus Sb and As) have been detected in brake linings and in
533 brake dust (Hulskotte et al., 2014; Grigoratos and Martini, 2015; Thorpe and Harrison, 2008).
534 However, certain elements stand out in brake-wear particles, either because they are in greater
535 abundance or because brake wear is a dominant source of these particles in ambient air and
536 therefore they act as useful tracers for brake-wear emissions. The four elements most readily
537 identified in brake-wear particles are Fe, Cu, Ba and Sb, which also tend to correlate well with each
538 other (e.g. Gietl et al., 2010; Lawrence et al., 2013) indicating a common source. Formerly, Pb was
539 also a significant component of brake wear but this is no longer the case due to the replacement of
540 Pb in modern linings (Grigoratos and Martini, 2015).

541

542 For Cu, Ba and Sb, brake wear emission is now their major source into ambient air, away from
543 specific industrial point source emissions (Hjortenkrans et al., 2007). For example, the UK National
544 Atmospheric Emissions Inventory (NAEI) estimates that, in 2016, 90% of UK national emissions of

545 Cu derived from non-exhaust emissions, a percentage that will be higher still in areas with large
546 road traffic activity. This is consistent with the 80-90% estimates made for the UK by Denier van
547 der Gon et al. (2007). Brake-wear emissions are also an important source of atmospheric emissions
548 of Sb (Bukowiecki et al., 2009; Iijima et al., 2008; Varrica et al., 2013): many brake linings contain
549 up to a few % of stibnite (Sb_2S_3), which can be oxidised to Sb_2O_3 during the braking process.
550 The three elements Cu, Ba and Sb, have all been used as chemical tracers for brake wear in source
551 apportionment studies (Alves et al., 2020; Amato et al., 2010a; Charron et al., 2019; Dong et al.,
552 2017; Dongarrà et al., 2008; Gietl et al., 2010; Hicks et al., 2021; Lawrence et al., 2013; Lough et
553 al., 2005; Pant et al., 2015). Where a mass ratio of the element to brake-wear particles can be
554 estimated or assumed, then quantification of that element in ambient air provides a quantitative
555 measure of the contribution of brake-wear emissions at the measurement location. For example,
556 Gietl et al. (2010) used Ba as a tracer for the contribution of brake-wear particles to different size
557 fractions of ambient PM at a roadside and an urban background site in central London. These
558 authors estimated that the mean proportion of Ba in brake wear emissions in PM_{10} (for the UK, at
559 that time) was 1.1%. Hicks et al. (2021) also used Ba as a tracer, applying a scaling factor of 91 to
560 their roadside increment in Ba concentration to derive the contribution to PM_{10} from brake wear.
561 Charron et al. (2019) report that ratios involving copper (Cu/Fe and Cu/Sn) could be used as brake-
562 wear emissions tracers as long as brakes with Cu remain in use.

563

564 Other studies that applied multivariate statistical techniques to a suite of PM elemental composition
565 data have likewise identified Cu, Ba and Sb (together with Fe) as associated with a brake-wear
566 source (e.g. Beddows et al., 2015; Crilley et al., 2017; Visser et al., 2015). (The multiple sources of
567 Fe in airborne PM mean it is not so useful as a source tracer.) In a harmonised study of five major
568 southern European cities, Amato et al. (2016a) identified by PMF a factor showing broadly similar
569 composition within all of the cities, separate to a vehicle exhaust factor, which they attributed to
570 vehicular non-exhaust emissions. The factor showed chemical features typical of brake wear (Cu,

571 Ba and Sb), tyre wear (Zn) and road dust (Si, Al and Ca). Since these elements do not derive
572 exclusively from one source, there are ambiguities and uncertainties in quantification of individual
573 non-exhaust emissions sources by these statistical approaches. It is also not possible unambiguously
574 to distinguish from the composition of ambient PM between direct emissions of brake (and tyre and
575 road) wear particles and the vehicle-induced resuspension or entrainment of brake dust deposited on
576 the road surface or road verges.

577

578 Compared with elemental abundances, there is very limited information concerning the presence of
579 organic compounds in brake wear particles (Grigoratos and Martini, 2015). This reflects a number
580 of factors: organic compounds are in lower abundance than metallic and mineral compounds in
581 brake linings and brake wear; quantification of individual organic species is analytically more
582 challenging than for individual elements; and the existence of elemental markers for brake wear
583 reduces the motivation to undertake speciation of organic compounds. Emissions of organic
584 compounds during braking derive from the high-temperature and high-pressure reactions of
585 graphite and phenolic resin components. Plachá et al. (2017) report BTEX, PAH and other semi-
586 volatile organic compound emissions from a low-metallic brake lining tested using the ISO 26867
587 friction evaluation standard procedure. Rogge et al. (1993) also reported PAHs in brake dust, with
588 their most abundant class of organic compounds being polyalkylene glycol ethers and n-alkanoic
589 acids. However, brake liner formulations will have changed since these earlier measurements were
590 made. In a recent study, Alves et al. (2021) report the presence of aliphatic and aromatic
591 hydrocarbons, alcohols, glycol/glycerol derivatives, plasticisers, sugars, sterols and various phenolic
592 constituents in particles derived from brake wear. Total carbon accounted for 5 – 74 % of PM₁₀
593 mass in tests on a range of pad formulations and braking cycles. A useful organic marker of brake-
594 wear particles is considered unlikely (Grigoratos and Martini, 2015; Thorpe and Harrison, 2008).
595 Non-frictional, regenerative braking systems significantly reduce this source of non-exhaust particle
596 emissions.

597 4.2 Tyre Wear Particles

598 In contrast to brake systems, whose composition is dominated by metals and metal-containing
599 inorganic compounds (such as silicates, sulphides, oxides), tyre composition is dominated by
600 organic materials. However, similar to brake systems, the specific composition of tyres varies
601 widely between vehicle type (e.g. passenger car, heavy-duty vehicle) and manufacturer. The main
602 categories of materials in a tyre by mass are as follows (Baensch-Baltruschat et al., 2020;
603 Grigoratos and Martini, 2014; Sommer et al., 2018).

- 604 • Rubber hydrocarbons (40-60%): natural rubber (polyisoprene $[C_5H_8]_n$) and synthetic rubber,
605 e.g. styrene butadiene rubber (SBR) or butadiene rubber (BR).
- 606 • Filler (20-35%): typically soot/carbon black, silica (SiO_2) and chalk ($CaCO_3$).
- 607 • Softener (15%): hydrocarbon oils and resin.
- 608 • Textile and metal reinforcement (5-10%).
- 609 • Vulcanization agents (2-5%): sulphur and zinc oxide (ZnO).
- 610 • Other additives (5-10%): preservatives (halogenated cyanoalkanes), anti-oxidants (amines,
611 phenols), desiccants (calcium oxides), and plasticizers (aromatic and aliphatic esters).

612

613 Although tyre wear is primarily carbonaceous, a number of inorganic materials are used in tyre
614 manufacture. In particular, ZnO is used as an activator for the vulcanisation process and Zn can
615 account for around 1% of tyre tread material (Smolders and Degryse, 2002; Davis et al., 2001).

616 Tyre wear is therefore a significant source of Zn in the environment (Councell et al., 2004; Adachi
617 and Tainosho, 2004; Blok, 2005). In the UK, the NAEI estimates that 23% of Zn emissions to air in
618 the UK in 2016 derived from tyre wear. This proportion will be considerably higher near roads.

619 Enrichment of Cd in tyre wear has also been observed (Fabretti et al., 2009; Hjortenkrans et al.,
620 2007; Lanzerstorfer, 2018). However, Cd is also a constituent of brake wear (Song and Gao, 2011),
621 although emissions from this latter source may be declining (Hjortenkrans et al., 2007), no Cd was
622 detected in tyre-wear particles by Kreider et al. (2010).

623 The significant enrichment of Zn in tyre-wear particles compared with crustal dust (e.g. Gustafsson
624 et al., 2008) has led to its routine use as a marker for tyre-wear emissions in the same way as Cu, Sb
625 and Ba are used as markers for brake wear in ambient PM source apportionment studies such as
626 those described above. For example, Hicks et al. (2021) applied a scaling factor of 50 to their
627 roadside increment in Zn concentration (taken from Harrison et al. (2012)) to derive the
628 contribution to PM₁₀ from tyre wear. However, as noted above, these markers are not unambiguous
629 for their respective sources and the suitability of Zn, in particular, has been discussed (Thorpe and
630 Harrison, 2008). Zn also has a significant anthropogenic source from wear of Zn-galvanised
631 materials such as fencing, and a source in biomass burning emissions.

632

633 In respect of other inorganic components in tyres, SiO₂ is commonly used as a reinforcing filler
634 (Rattanasom et al., 2007) and a prevalence of Si-rich particles has been observed from tyre-wear
635 driving simulators (Kim and Lee, 2018; Park et al., 2018). These particles are particularly prevalent
636 in the ultrafine fraction (Park et al., 2018), whose production is substantially enhanced when tyre-
637 tread surface temperatures reached 160°C (Park et al., 2017). Sulphur was identified in the larger
638 particles (Park et al., 2018), but S is not useful as an elemental marker of tyre-wear in ambient PM
639 because of the prevalence of sulphate (SO₄²⁻) from the oxidation of SO₂ emissions. Black carbon is
640 also a significant component of tyre-wear emissions – the EMEP/EEA Emissions Inventory
641 Guidebook applies a BC/PM_{2.5} ratio of 0.15 for tyre wear (EMEP/EEA, 2016) – but the BC
642 emission is both highly uncertain and not a unique source marker for tyre wear.

643

644 Information on the organic compounds in tyre wear particles is sparse because it is laborious to
645 extract, separate and quantify individual organic species by GC-MS, LC-MS and similar off-line
646 techniques (Rogge et al., 1993; Nolte et al., 2002). The focus in this review is identification of
647 potential organic markers for a tyre-wear source. One such class of compounds is benzothiazole
648 (BT) and its derivatives (Rogge et al., 1993; Lin et al., 2010; Zhang et al., 2018), which are used as

649 vulcanization accelerators and UV-light stabilisers for rubbers (Liao et al., 2018). Zhang et al.
650 (2018) quantified benzothiazole and six of its derivatives in tyre-wear particles using solid-liquid
651 extraction and ultra-high performance liquid chromatography-tandem mass spectrometry (UPLC-
652 MS/MS). All seven BTs were found in 17 tyre-wear particle samples from different tyre brands.
653 The mass fractions of all seven BTs summed ranged from 46.93 to 215 $\mu\text{g g}^{-1}$ with an average
654 concentration of 99.32 $\mu\text{g g}^{-1}$. Benzothiazole and 2-hydroxybenzothiazole (2-OH-BT) were the two
655 major compounds, accounting for 56-89% of the total.

656

657 A second class of potential organic markers for tyre wear emissions are hydrogenated resin acids.
658 Resin acids are natural products found in pine tars, which are additives used by manufacturers to
659 give tyres certain desirable properties (Nolte et al., 2002). Some of these resin acids are chemically
660 hydrogenated before being added during the manufacture process, which differentiates them from
661 the natural resin acids that may be present in ambient PM from sources such as wood smoke.
662 Kumata et al. (2011) proposed the following four dihydroresin acids as molecular markers for tyre-
663 wear emissions: 8-isopimaren-18-oic acid (I), 8-pimaren-18-oic acid (II), 13 β (H)-abieten-18-oic
664 acid (III), and 13 α (H)-abiet-8-en-18-oic acid (IV). The absence of I–IV in natural sources and the
665 linear correlations between dihydroresin acids with different skeletons in tyres and in environmental
666 samples demonstrated that I–IV are specific markers for synthetic rubbers. These workers further
667 suggest from the results of photolysis experiments that I–IV can set lower limits for tyre-wear
668 contributions to environmental loads of PM and polycyclic aromatic hydrocarbons (PAHs) with
669 molecular weight ≥ 202 . By comparing III + IV concentrations or (III+IV)/pyrene or
670 (III+IV)/benzo[a]pyrene ratios in tyres and those in environmental matrices, Kumata et al. (2011)
671 estimated the contributions of tyre-wear emissions to airborne PM, pyrene, and benzo[a]pyrene as,
672 respectively, $0.68 \pm 0.54\%$, $6.9 \pm 4.8\%$, and $0.37 \pm 0.18\%$ in roadside PM and $0.83 \pm 0.21\%$, $0.88 \pm$
673 0.52% , and $0.08 \pm 0.06\%$ in rooftop PM.

674

675 Other studies have also highlighted PAHs in tyre-wear emissions. In an experimental simulation of
676 tyre-road interaction, Aatmeeyata and Sharma (2010) observed a linear increase in cumulative
677 emissions of PAH (in particles of diameter > 10 μm) over a 14,500 km run of the tyre. The most
678 abundant PAH was pyrene ($30 \pm 4 \text{ mg kg}^{-1}$), followed by benzo[ghi]perylene ($17 \pm 2 \text{ mg kg}^{-1}$).
679 Sadiktsis et al. (2012) analysed for 15 high molecular weight PAH in 8 different tyres, albeit not in
680 tyre-wear emissions. Whilst there was a factor 20 difference in PAH concentration between tyres,
681 the relative abundance of PAHs was quite similar. Almost all (92%) of the total extractable PAH
682 content was attributed to five PAHs: benzo[ghi]perylene, coronene, indeno[1,2,3-cd]pyrene,
683 benzo[e]pyrene, and benzo[a]pyrene. The four dibenzopyrene isomers analysed
684 (dibenzo[a,l]pyrene, dibenzo[a,e]pyrene, dibenzo[a,i]pyrene, dibenzo[a,h]pyrene) constituted <2%
685 of the sum of the 15 PAH determined but represent a potential hazard as they are carcinogenic. In
686 the EU, the concentrations of PAH in tyre-wear emissions should have declined markedly from
687 2010 due to implementation of EU Directive 2005/69/EC that prohibits the use of 'PAH-rich'
688 extender oils in tyres produced after January 2010 (AQEG, 2019). It is also important to note that
689 PAH are emitted to the air from multiple sources other than via tyre-wear particles, in particular for
690 the road environment, in diesel exhaust emissions.

691

692 ATOFMS presents the opportunity to seek m/z distributions unique to a particular source in the
693 mass spectra of individual particles (Dall'Osto et al., 2014). ATOFMS mass spectra of tyre-wear
694 particles derived in a laboratory simulator showed peaks due to sulphur compounds, nitrate, Zn and
695 m/z values >100 attributed to organic polymers for particles in the size range 200–3000 nm.

696 Examination of large datasets of ATOFMS spectra for road dust and PM in ambient air revealed
697 that the majority (95%) of tyre-wear particles present in the road dust samples and atmospheric
698 samples were internally mixed with metals (Li, Na, Ca, Fe, Ti), as well as phosphate.

699

700 The extent to which tyre-wear (and road-wear) particles contribute to the microscopic plastic
701 particles (microplastics) entering the environment depends on the definition of plastic. There is
702 some disagreement on which polymers are “plastics”. The ISO 472:2013 definition of plastic is
703 “material which contains as an essential ingredient a high molecular weight polymer and which, at
704 some stage in its processing into finished products, can be shaped by flow” (Hartmann et al., 2019).
705 Rubbers (and some other elastomers) are excluded from this definition of plastic. However, this
706 definition reflects the historic industrial landscape rather than perspectives about the behaviour of
707 rubber fragments in the environment. Other authors include rubber within their definition of plastics
708 and conclude that tyre-wear is a substantial, and under-reported, source of microplastics into the
709 environment (e.g. Kole et al., 2017; Knight et al., 2020; Sommer et al., 2018). The annual global
710 rubber production of ~27 million tonnes in 2015 was ~9% the total plastic production (Kole et al.,
711 2017). However, estimating the fluxes of these plastic and rubber materials in the form of
712 microparticles into the environment is a challenge. A proportion of tyre wear will not be released to
713 air but remain on the road surface or enter the environment in run-off. A further consideration in
714 terming tyre wear as plastic arises from the way in which tyre wear particles may be substantially
715 (50%) internally mixed with road-wear fragments (Kreider et al., 2010; Panko et al., 2013; Panko et
716 al., 2019).

717

718 **4.3 Road Surface Wear Particles**

719 It is not possible from field measurements to distinguish unambiguously between direct emissions
720 of road surface wear particles and resuspension of such particles already on the road surface since
721 the chemical composition is often very similar (Bukowiecki et al., 2010). Using mobile load
722 simulators to compare the two contributions, Gehrig et al. (2010) concluded that road abrasion
723 fluxes are normally lower than re-suspension fluxes, and that the latter increase with the dust
724 loading on the road, as expected. However, road-surface wear emissions are substantially enhanced

725 in locations with sub-zero temperature winters where studded tyres are routinely used for additional
726 traction (Denby et al., 2018; Gustafsson, 2018).

727

728 Road surfaces are broadly composed of concrete or asphalt/tarmac. The former is a mixture of
729 mineral aggregate, sand, and cement. The latter comprises ~95% mineral aggregates and ~5%
730 bituminous binder. Since the dominant component in both concrete and asphalt road surfaces is the
731 aggregate, road-wear particles mainly consists of small mineral fragments whose composition
732 varies with the vast array of geological sources of the aggregate. The dominant crustal elements in
733 these minerals are Si, Ca, K, Fe, and Al. Whilst a grouping of these elements is useful to quantify
734 road dust contribution to airborne PM (see next section), it is nearly impossible to separate primary
735 road wear from other mineral dust deposited on the road (Denier van der Gon et al., 2013).

736

737 The bituminous binder in asphalt/tarmac road surfaces comprise many thousands of high molecular
738 weight aliphatic and aromatic hydrocarbons, which may also contain heteroatom functional groups
739 with O, N and S. Smaller amounts of polymers, epoxy resins and metal complexes are also present
740 to enhance binder-aggregate bonding (NIOSH, 2000; Thorpe and Harrison, 2008). There is a
741 paucity of detailed chemical information on road surface composition, but it is unlikely that there is
742 a definitive molecular (or elemental) marker for road surface wear (Boulter, 2005; Thorpe and
743 Harrison, 2008). Many studies have reported elevated concentrations (0.08–149 $\mu\text{g/g}$) of PAHs in
744 urban road dusts (Hwang et al., 2019), and the much greater prevalence of PAHs in road dust from
745 coal-tar-based compared with asphalt-based road surfaces indicates their predominant source is
746 road-surface wear (Metre et al., 2009) (coal-based bitumen containing substantially greater PAH
747 content than oil-based bitumen). However, coal-based tars are now infrequently used.

748

749

750

751 **4.4 Resuspension Particles**

752 The traffic-related resuspension source refers to the uplift into the air by vehicle movement of
753 particles already present on the road surface, mediated for example via tyre shear stress or vehicle
754 wake turbulence. Resuspended particles therefore comprise particles from all three of the sources
755 already described – brake wear, tyre wear, road surface wear – as well as particles from other
756 sources that have also deposited onto the road surface, for example exhaust emission particles,
757 particles from de-icing and gritting, wind-blown dust and biogenic particles. Consequently, the
758 composition of resuspension particles comprise all the chemical components already described, and
759 many more. Chemical analysis of road dust has received particular attention because of its
760 contribution to potential toxicity to near-roadside ecosystems and to aquatic systems via run-off as
761 well as to inhaled airborne PM. The very wide range of elemental and organic components in road
762 dust is highlighted in a number of recent reviews (Haynes et al., 2020; Hwang et al., 2016; Hwang
763 et al., 2019; Khan and Strand, 2018; Lanzerstorfer, 2020; Maeaba et al., 2020). Individual studies
764 identify mutual correlation and/or enrichment in road dust particles of the ‘crustal’ elements
765 associated with road surface mineral aggregates and wind-blown dust, for example Si, Al, Ca, Fe,
766 Ti and Mg (Alves et al., 2020; Amato et al., 2009a; Amato et al., 2011; Amato et al., 2012a;
767 Dall'Osto et al., 2013; Fabretti et al., 2009; Ondráček et al., 2011; Padoan et al., 2017).

768

769 Road dust is also highly enriched and contaminated by elements from tyre and brake wear that have
770 deposited onto the surface, e.g. Sb, Cu, Zn, Sn, Cr (Adamiec et al., 2016; Alves et al., 2020; Cowan
771 et al., 2021; De Silva et al., 2021; Lanzerstorfer, 2018; Padoan et al., 2017; Pipalatkar et al., 2014).
772 A class of elements in road dust particles that has received particular attention due to their toxicity
773 (and potential for recovery) are the platinum group elements Pt, Pd, Rh derived from wear of
774 exhaust after-treatment catalysts (De Silva et al., 2021; Ladonin, 2018; Wang and Li, 2012; Zereini
775 et al., 2012). Whilst the ultimate source of these elements is exhaust emissions, when particles

776 containing these elements deposit on the road surface and are resuspended they contribute to the
777 chemical composition of traffic-derived non-exhaust particles.

778

779 **5. MEASUREMENTS OF NEE IN THE ATMOSPHERE**

780 **5.1 Principles of Measurements**

781 NEE particles are, by their nature, refractory and insoluble in water. This means that many of the
782 common tools for analysing particulate composition (both offline and in situ) such as ion
783 chromatography, thermal desorption gas chromatography and thermal aerosol mass spectrometry
784 such as the aerosol chemical speciation monitor (ACMS), are not well suited for analysis. Instead,
785 offline analysis methods usually focus on the analysis of the metallic fraction. A common method is
786 the digestion of a sample in acid (usually nitric acid) followed by analysis using Inductively
787 Coupled Plasma (ICP) analysis with Optical Emission Spectroscopy (OES, also known as Atomic
788 Emission Spectroscopy or AES) or Mass Spectrometry (MS) (Chow and Watson, 2013; Chow,
789 1995). It is also common to analyse metals in place on a substrate using x-ray spectroscopic
790 methods such as X-Ray Fluorescence (XRF). To increase sensitivity, this may use a synchrotron
791 source (Bukowiecki et al., 2005). The particles can also be analysed using electron microscopy and
792 classified according to morphology and composition (Weinbruch et al., 2014).

793

794 The selective analysis of the carbonaceous component of tyre wear that makes up the majority of its
795 composition is analytically challenging because it is (by design) not volatile or soluble, so cannot
796 generally be analysed using thermal desorption or liquid chromatography (Noziere et al., 2015).
797 However, it is possible to detect certain organic tracers and toxins such as PAHs and plasticisers
798 (Avagyan et al., 2014; Alves et al., 2018). Another method to study the organic fraction of tyre
799 material is pyrolysis, followed by Gas Chromatography (py-GC), a technique that has been
800 successfully used to detect the presence of tyre wear particles in groundwater and soils (Unice et al.,
801 2012). It is also possible to use FTIR to study the polymers present (Mengistu et al., 2019).

802 Another feature of NEE particles that is distinct from many other atmospheric aerosols is their size,
803 owing to the fact they are produced mainly from abrasion rather than combustion or chemical
804 processes. This means that in terms of mass, they tend to occupy predominantly the coarse $PM_{2.5-10}$
805 fraction rather than the $PM_{2.5}$ fraction. This is evident when samples are obtained using size-
806 segregated methods such as cascade impactors (e.g. Kwak et al., 2013). NEE particles can also be
807 associated with modes identified in the data from aerosol size spectrometers that are optimised for
808 micron sized particles, such as the Aerodynamic Particle Sizer (APS) and optical particle counters
809 (Harrison et al., 2012). By way of example, Harrison et al. (2011) reported a volume concentration
810 size mode at $2\ \mu m$ for brake dust and at $>10\ \mu m$ for resuspended road dust. By contrast, the mode
811 corresponding to exhaust emissions existed at $0.2\ \mu m$. Also, Kreider et al. (2010) reported volume
812 modes of around $70\ \mu m$ for roadway particles and $100\ \mu m$ for tyre wear particles.

813

814 It is known that NEE can be a source of ultrafine particles, i.e. those smaller than 100 nm (Kwak et
815 al., 2014; Nosko et al., 2017). These contribute little to the PM mass concentrations, but can
816 dominate number concentrations. While it is possible to measure these in the ambient atmosphere
817 using mobility size spectrometers and condensation particle counters (Wiedensohler et al., 2012),
818 exhaust emissions are known to be a major source of these, so disentangling the NEE contribution
819 is challenging.

820

821 Semi-continuous analysis of these particles is possible using in situ XRF analysis, as used in the
822 XACT monitor (Tremper et al., 2018; Furger et al., 2017). This has the advantage over offline
823 analysis in that it produces data in near real time and can produce data on an hourly basis. Real-time
824 analysis is also offered by single particle mass spectrometers such as the ATOFMS and related
825 instruments (Murphy, 2007; Pratt and Prather, 2012). These use pulsed lasers to vaporise and
826 ionise the components of individual particles and analyse the resultant ions using mass
827 spectrometry. This can identify particle types through numerical clustering of the mass

828 spectrometric data (Rebotier and Prather, 2007). This can assess the relative contributions of NEE
829 particles to ambient number concentrations (Beddows et al., 2016; Dall'Osto et al., 2014). Because
830 these instruments are usually operated using optical detection to trigger the desorption later, the data
831 is limited to particles of around 200 nm or greater.

832

833 **5.2 Receptor Modelling Techniques**

834 The influence of individual sources of particulate matter at a receptor site can be qualitatively
835 inferred from the inspection of tracers (Gietl et al., 2010). This works particularly well if a tracer is
836 unique to a particular source, so in the case of NEE, tracers such as Sb and Cu can be taken as
837 indicative of brake wear, but depending on the environment, it is harder associating tracers such as
838 zinc with tyre wear when it is also associated with wood burning. These can be compared with data
839 from other sources to estimate contributions to the overall particulate budget (Harrison et al., 2012).
840 The ISO/TS 20593:2017 standard covers the estimation of tyre and road wear contributions based
841 on analysis of samples using pyrolysis.

842

843 A common method of aerosol receptor modelling for the purposes of quantitative source
844 apportionment as applied to other aerosol types is the Chemical Mass Balance (CMB) technique,
845 which involves estimating a source's contribution to PM through inspection of tracers and
846 application of knowledge of the emissions from a source (Chow and Watson, 2002). However, this
847 does not work particularly well for NEE because the source profiles are known to vary greatly
848 according to formulation in the case of brake wear and the local composition in the case of road
849 dust.

850

851 As an alternative, non-negative factorisation techniques are often applied, in particular PMF
852 (Paatero, 1997; Paatero and Tapper, 1994). This method is able to deconstruct a multivariate dataset
853 into a specified number of 'factors' using error-weighted residual minimisation without any a priori

854 assumptions regarding the nature of the factors. This method works very well in source
855 apportionment when a given contribution's profile is invariant (has a fixed source profile that is not
856 modified in the atmosphere) and can be applied to a composite of observations, such as the time
857 series of multiple metals derived using ICP-MS analysis. The technique is further improved when
858 data is obtained as a function of time of day, thereby capturing variations in traffic density, so this
859 technique is especially powerful when applied to high frequency sampling and semicontinuous
860 analysis. PMF can also be applied to size distribution data, which can also be used to resolve
861 different sources (Harrison et al., 2011; Vu et al., 2015).

862

863 A limitation of XRF is that it struggles to separate sources that are covariant at a given receptor site,
864 which can frequently be the case with NEE, so sometimes these are lumped together as a single
865 'traffic related' factor. Moreover, there is a conceptual problem with this data model when it comes
866 to separating road dust from brake and tyre wear based on chemical measurements, because much
867 of the road dust is contaminated with the chemical components of the other sources (Alves et al.,
868 2018). Likewise, tyre wear particles are also known to contain small road dust particles (Kreider et
869 al., 2010). The only receptor measurements theoretically capable of distinguishing these are the
870 single particle methods such as electron microscopy and single particle mass spectrometry, however
871 these are still not able to explicitly tell whether a given particle was emitted directly to the
872 atmosphere, or initially settled to be subsequently resuspended.

873

874 A further variation of PMF is the application of the Multilinear Engine (ME2) algorithm, which
875 through adding additional penalty functions to the weighted residuals, enables the user to invoke a
876 degree of a priori knowledge, by assuming 'target' profiles or 'pulling' time series towards a
877 desired solution (Amato and Hopke, 2012; Paatero, 1999; Amato et al., 2009b). Another expansion
878 of the PMF technique includes bringing in data from multiple sources; if (say) metal data alone
879 were to be used, the derived factors would only concern the contributions to the available metal data

880 in isolation, but by including other chemical analyses, size spectrometer and/or overall particulate
 881 mass data, the contributions to the PM mass budget can be estimated.

882

883 **5.3 Measured Contributions and Size Distributions**

884 A common method of capturing NEE data is by assessing its contribution to the roadside increment,
 885 i.e. comparing roadside concentrations with equivalent background measurements for a given
 886 conurbation. This can be done for mass concentrations of particulate components and size resolved
 887 data. NEE has been seen as a responsible for a general roadside or kerbside increment in PM_{2.5-10}
 888 and this can be linked to metals associated with NEE (Hopke et al., 2018; Lenschow et al., 2001;
 889 Querol et al., 2008; Lough et al., 2005; Gietl et al., 2010). Road dust can also be sampled directly
 890 using suction and aerodynamic separation, to obtain a sample representative of what could
 891 contribute to ambient PM₁₀ if resuspended by traffic (Amato et al., 2016b; Alves et al., 2020). This
 892 has the advantage of providing a larger volume of material for analysis.

893

894 As shown in Table 2, various metals can be associated with different NEE sources based on
 895 ambient measurements. The associations with specific sources can be based on knowledge of
 896 emissions, analysing the roadside increment and tunnel studies. Some elements can be associated
 897 with multiple sources such as iron present in both brake wear and mineral dust. A common method
 898 of inspecting the brake wear contribution is through evaluation of the Cu:Sb ratio, and values
 899 between approximately 1 and 10 have been reported (Pant and Harrison, 2013, and references
 900 therein).

901

902 **Table 2:** Metal profiles of various NEE and mineral dusts based on selected ambient, tunnel and
 903 road dust studies. *Metals detected at a subset of sites in parenthesis.

904

Reference	Brake wear	Tyre wear	Combined vehicular	Mineral dust
Tanner et al. (2008)	Cu, Cd	Zn		
Fabretti et al. (2009)			Cu, Zn, Sb, Sn	Rb, Sr, Mn, Fe, As
Gietl et al. (2010)			Ba, Fe, Cu, Sb	
Pérez et al. (2010)			Sb, Cu, Ni, Sn	Fe

Amato et al. (2011)	Fe, Cu, Zn, Cr, Sn, Sb	Zn		Al, Ca, Fe, V
Harrison et al. (2012)	Ba, Cu, Fe, Sb	Zn		Si, Al
Visser et al. (2015)	Cu, Zr, Sb, Ba		Fe	Si, Ca
Amato et al. (2016b)	Cu, Sn, Sb, Cr, Ba, Fe, Al, Ca	Ca		Al, Ca, K, Ti, Fe, Mg
Crilley et al. (2017)			Ti, Cr, Mn, Fe, Ni, Cu, Rb, Zr, Mo	Al, Si, Ca, Ti, Mn, Ni, Zn
Alves et al. (2018)			Pb, An, Fe, Cu, Sn, Sb	Al
Jeong et al. (2019)	Cu, Ba			Ca
Alves et al. (2020)			Cu, Zn, Sn, Sb, As, Se, Bi, Cd, Pb	Si, Al, Fe, Ca, K
Daellenbach et al. (2020)			Cu, Fe, Mo, Sb	Al, Se, Ti
Rai et al. (2020)			Fe, Cr, Cu, Mn, Zn, Ba	Ca, Si
Zhang et al. (2020)	Fe, Ba, Zr, Cu, Ti	Si, Zn		Si, Ca
Rai et al. (2021)			Ba, Ni, Mn (Rb, Cr, Fe, Zr, Sr)	Ca, Si, Ti (Sr, Fe, Zr)

905

906 Mineral dust can be associated with Si, Al and Ca, but it is can be harder to unambiguously identify
907 because its composition varies with local mineralogy, nearby sources (e.g. construction) and
908 contamination from exhaust and non-exhaust emissions (Adamiec et al., 2016). In some works, the
909 contribution of traffic to sampled material is expressed as an ‘enrichment’ relative to the local
910 continental crust (using an element such as Al as a reference), thereby capturing all the non-natural
911 components (Alves et al., 2018). Note that this does by no means imply that airborne mineral dust is
912 inherently ‘natural’ as the vehicles are still responsible for its resuspension. In addition, a
913 component of road salt is sometimes reported in the form of Na, Mg and Cl (not shown in table).

914

915 Other components of the brake and tyre particles include elemental and organic carbon as measured
916 using thermal-optical analysis (Amato et al., 2016b), however in isolation, these are difficult to
917 distinguish from exhaust emissions, which are also predominantly carbonaceous in nature. The
918 polymer markers associated with tyre wear, as analysed using py-GC, are relatively unambiguous
919 and can be used to assess its contribution (Panko et al., 2013).

920

921 The application of PMF to urban data has allowed the emissions to be profiled beyond individual
922 elements and complete profiles for the different components of NEE as observed at a receptor site,
923 although as explained above, it is often difficult to separate the brake and tyre wear components.
924 Examples include Fabretti et al. (2009), who were able to divide $PM_{2.5}$ into three categories,
925 specifically resuspension, a combined ‘vehicle abrasion’ factor and exhaust emissions. Another
926 example is the work of Rai et al. (2020), who were able to separate ‘road dust’ and ‘traffic related’
927 factors from background sources and fireworks in XACT data. Further insight can also be gained
928 through the comparison of data collected at multiple sites simultaneously. Through PMF analysis of
929 size-resolved metal data from roadside, urban background and rural measurements in London,
930 Visser et al. (2015) were able to profile and assess the urban and roadside increments due to
931 resuspended dust, brake wear, road salt and other traffic related aerosols. This again found that NEE
932 could be most associated with PM_{10} , but an increment in $PM_{2.5}$ was also detected.

933

934 Some studies have presented a combined factorisation analysis from multiple data sources. For
935 instance, Crilley et al. (2017) were able to use metal data combined with size distributions.
936 Contributions of vehicle wear and road dust to $PM_{2.5-10}$ particles were found to be 13 and 31%
937 respectively at a roadside site, although the contributions to the fine fraction and a background site
938 were not disentangled from general traffic emissions. In a similar vein, Jeong et al. (2019) presented
939 a combined factorisation of $PM_{2.5}$ data obtained in Toronto, using a variety of online techniques
940 including an XACT, ACSM and Aethalometer, and offline analysis. This showed that NEE
941 represented around a third of traffic-related $PM_{2.5}$.

942

943 While many factorisation papers (e.g. Rai et al., 2020) report NEE as a percentage of the total
944 elements measured using a given analysis, these do not strictly represent contributions to the total
945 PM budget because of other components that are not measured during XRF or ICP analysis (e.g.
946 carbon, oxygen and hydrogen). Instead, tracers or a combined factorisation must be used.

947 The papers that do estimate absolute contributions (summarised in Table 3) have presented a
 948 significant contribution of NEE to coarse aerosol in urban environments. Based on tracer-derived
 949 data originally presented by Harrison et al. (2012), Beddows and Harrison (2021) reported the
 950 kerbside PM₁₀ increment at the London Marylebone Road site due to brake wear, tyre wear and
 951 resuspension to be 2.8, 0.5 and 1.5 $\mu\text{g m}^{-3}$ respectively. During a later study at the same site, Crilley
 952 et al. (2017) used combined factorisation to estimate the total contribution of vehicle wear and
 953 resuspension to PM_{2.5-10} to be 1.3 and 3.1 $\mu\text{g m}^{-3}$ respectively. Through the analysis and
 954 classification of samples analysed using electron microscopy, Weinbruch et al. (2014) estimated a
 955 PM₁₀ kerbside increment in Ruhr, Germany due to abrasion and resuspension of 0.8 and 4.2 $\mu\text{g m}^{-3}$
 956 respectively.

957

958 In contrast, estimates of the contribution to PM_{2.5} tend to be more minor. In Toronto, Jeong et al.
 959 (2019) used combined factorisation to estimate the average brake wear and resuspension
 960 contributions to be 0.2 $\mu\text{g m}^{-3}$ each at a downtown site and 0.6 and 0.3 $\mu\text{g m}^{-3}$ respectively at a
 961 highway site. While concentrations were low, it was stressed that these still represented the majority
 962 of some metals such as Ba, Cu and Mn. Using a tracer method applied to samples collected in
 963 London, Tokyo and Los Angeles, Panko et al. (2019) estimated that tyre wear particles represented
 964 only 0.1-0.68% of PM_{2.5} mass. Absolute concentrations varied, but the highest concentration of
 965 0.29 $\mu\text{g m}^{-3}$ was noted at the Blackwall Tunnel approach in London, where significant braking
 966 occurs.

967 Table 3: Summary of quantitative mass apportionment studies of NEE in polluted urban
 968 environments. Notes: (a) Kerbside increment rather than overall concentration reported; (b)
 969 Downtown site; (c) Highway site.

970

Reference	Brake Wear	Tyre Wear	Combined wear	Resuspension
PM₁₀ ($\mu\text{g m}^{-3}$)				
Beddows and Harrison (2021)	2.8 (a)	0.5 (a)		1.5 (a)
Crilley et al. (2017)			1.3	3.1
Weinbruch et al. (2014)			0.8 (a)	4.2 (a)
PM_{2.5} ($\mu\text{g m}^{-3}$)				
Jeong et al. (2019)	0.2 (b)			0.2 (b)

	0.6 (c)			0.3 (c)
Panko et al. (2019)		≤ 0.29		

971

972 **6. MODELLED CONCENTRATIONS OF NEE**

973 **6.1 Modelling Approaches and Data**

974 The usual purpose of modelling particulate concentrations is to predict annual-mean or daily-mean
975 total mass concentrations in locations where measurements have not been made, or under
976 alternative scenarios. Modelling thus refines estimates of exposure and allows interventions to be
977 tested. Local-scale models provide high output resolution (metres) close to roads (e.g. Hood et al.,
978 2014). Regional models provide coarser output resolution but cover much larger areas (e.g.
979 Simpson et al., 2003). Some models have multi-scale capabilities (e.g. USEPA, 2020b) and
980 different types of models may also be nested (e.g. Vieno et al., 2016). Regardless of the spatial
981 scale, emissions of NEE are usually calculated using the same information which underpins the
982 national inventories, as described in Section 2. These emissions are then aggregated with those
983 from vehicle exhausts for input to the model, meaning that their spatial distribution across road
984 links is closely aligned with that of exhaust emissions. This is not always appropriate, for example
985 the largest braking emissions may occur on road segments where accelerations and hence exhaust
986 emissions are relatively low. Most models of NEE concentrations are mechanistically separated
987 from the emissions calculations. For example, the treatment of local weather conditions for
988 dispersion is independent of any consideration of the impact of rainfall when calculating emissions.
989 Similarly, while traffic-induced turbulence can be varied in some models, this is not used to
990 calculate emissions. Where particle size distributions are defined (e.g. Archer-Nicholls et al., 2014)
991 they do not usually take explicit account of relative source-strengths of the different emission types:
992 exhaust, brake, tyre and road wear or resuspension. The relative contribution to concentrations
993 made by different traffic sources thus mirrors that defined by the emissions inventories.

994

995 Concentration modelling presents an opportunity to verify against ambient measurements and thus
996 test the effective performance of NEE emissions factors. Such comparisons are necessarily
997 confounded by uncertainties in the model to which the calculated emissions are input, and made
998 more difficult because of the relatively large contribution of non-traffic sources to roadside
999 concentrations when compared with NO_x. A common approach is to compare modelled and
1000 measured concentrations while varying the basis of emissions calculation (e.g. de la Paz et al.,
1001 2015).

1002

1003 Denby et al. (2013) used concentration modelling to test the performance of NORTRIP emissions
1004 calculations, which take specific account of winter salting and sanding, as well as road wear from
1005 studded tyres used in northern Europe. The model successfully predicted the effect on
1006 concentrations of a reduction in studded tyre use in Stockholm, but performed less well in
1007 Copenhagen, where studded tyres are not used at all. Inclusion of road salt within the emissions
1008 calculations led to a 6-16% increase in mean total concentrations and improved overall model
1009 performance.

1010

1011 Pay et al. (2011) used the CALIOPE model for Spain to compare EEA emissions factors with a
1012 bespoke routine for resuspension, which they derived from ambient measurements made elsewhere
1013 in Europe, adjusted for local rainfall. The bespoke resuspension calculations reduced, but did not
1014 remove, negative bias in the modelled concentrations. Resuspension was predicted to contribute up
1015 to 7 µg/m³ to annual mean PM₁₀ in urban areas. Amato et al. (2016c) predicted concentrations in
1016 Barcelona using a Gaussian line-source model, exhaust and wear emissions from TREMOVE 2.7,
1017 and an experimentally-derived fleet-average resuspension emission rate for Barcelona of 82 mg
1018 vkm⁻¹ which was reduced for periods during and after rainfall using a response curve determined
1019 from local measurements (Amato et al., 2012b). Applying this curvilinear relationship to rainfall
1020 improved model performance compared with turning resuspension on/off or assuming a constant

1021 emission. The bespoke resuspension emission routine removed most negative bias in predicted
1022 concentrations, but the spatially-uniform resuspension emission factor could not account for large
1023 inter-site variations. At traffic sites, resuspension contributed 23-44% of PM₁₀, with other traffic-
1024 related emissions contributing 8-14%. The importance of resuspension was highest in the most
1025 densely-populated parts of the city. Most modelling studies do not include rain frequency as part of
1026 the emissions calculations.

1027

1028 Hood et al. (2018) predicted annual mean PM₁₀ and PM_{2.5} concentrations across London using
1029 ADMS-Urban coupled with EMEP4UK. EEA emissions factors were adjusted based on
1030 measurements made in London (Harrison et al., 2012). AQEG (2019) used these results to show
1031 that NEE contributed more than exhaust emissions to annual mean PM₁₀ and PM_{2.5} and that NEE
1032 thus strongly impacted the predicted spatial variation in annual mean PM_{2.5} across central London.

1033

1034 Daellenbach et al. (2020) used a regional chemical transport model to calculate exposure across
1035 Europe to both PM mass and to its oxidative potential (a measure of the ability to deplete anti-
1036 oxidant defences in the body). NEE emissions were not explicitly represented, but were included as
1037 a function of NO_x emissions, with size distributions subsequently parameterised from ambient
1038 measurements. The model results compared well with measurements and were used to inform a
1039 wider discussion of the oxidative potential of PM.

1040

1041 Machine learning methods may also be used to predict total PM mass concentrations at individual
1042 monitoring sites under alternative traffic management scenarios (e.g. Suileman et. al, 2019). These
1043 approaches have similarities to source receptor modelling and, while not widely used in this
1044 context, have the potential to determine the site-specific dependence of total concentrations on
1045 individual causal factors, such as rainfall or traffic volume (Carslaw and Taylor, 2009); potentially

1046 allowing alternative scenarios to be tested without resolving the NEE contribution and without
1047 reliance on mechanistic emissions factors.

1048

1049 **7. MITIGATION MEASURES**

1050 **7.1 Engineering Control and Material Reformulation**

1051 There has been consistent effort by automotive manufacturers and aftermarket parts manufacturers
1052 to develop brake systems which have good wear characteristics and good characteristics with
1053 regards fouling of nearby surfaces. The drivers have historically related to cleanliness during
1054 operation (e.g., Gaylard et al., 2010) and controlling operating costs, however, the compromise with
1055 braking performance (including noise, friction characteristics and thermal behaviour) has meant that
1056 brake particulates have not had the focus they are now receiving.

1057

1058 Attention was drawn around 2006 to the contribution of brake pads on the aquatic ecosystem in the
1059 San Francisco Bay area, US as a result of emissions of copper containing particles (e.g., Rosselot,
1060 2006). This attention brought into focus the need for reduction of copper emissions from brake
1061 systems and led to a marked increase in commercial interest in reducing particulate emissions from
1062 brakes evident in the proliferation of related patents at this time. A memorandum of understanding
1063 between US EPA and the automotive industry was signed in 2015 with the aim of phasing out the
1064 use of copper in brake pads (to less than 0.5% by weight) by 2025 (Memorandum of Understanding,
1065 2015), which may itself resolve the copper issue, but contributed to the awareness of brake systems
1066 as sources of vehicular particulate emissions. The studies investigating technological approaches to
1067 emission reduction or control therefore remain of interest.

1068

1069 There are two technological approaches evident in the literature for reducing airborne particulate
1070 emissions from brakes, namely changing the formulation of the brake friction materials and adding

1071 technology to trap the particles before they are released into the surroundings. It can be inferred also
1072 that vehicle design decisions beyond the friction brake system will influence particle emissions.

1073

1074 **7.2 Friction Material Formulation**

1075 New materials have been evaluated thoroughly in respect of friction characteristics, noise and wear
1076 rates, but only recently has significant attention focused on particle emissions. Low copper
1077 formulations developed in light of the MoU have often focused on wear rather than emissions
1078 limiting the amount of data on the effects of replacing copper fibres in brake pads with alternative
1079 metallic materials (commonly steels), and non-metallic materials (such as natural fibres and
1080 ceramics). Despite the expectation of mass balance, wear rates typically show a weak correlation
1081 with particle number emissions at lower brake material temperatures (Tarasiuk et al., 2020) and are
1082 therefore not a reliable indicator.

1083

1084 There are a myriad of options available to manufacturers when developing new friction materials
1085 such as the materials used to create the primary plateaus and the binders and other materials which
1086 contribute to the secondary plateau formation. Despite only a fraction of this design space having
1087 been explored in the literature, it is clear that brake material formulation has a significant effect on
1088 particle emissions both in laboratory environments (e.g. Lyu et al., 2020) and on road (Farwick zum
1089 Hagen et al., 2019a).

1090

1091 Ceramic fibre and steel fibre based friction materials have been shown to produce higher particle
1092 number emissions than copper based materials at lower operating temperatures (<100°C) (Wei et
1093 al., 2020) raising concerns about the risk of particle emissions increasing as a result of reducing
1094 copper content. It is well recognised that a transition temperature exists at which the resins in the
1095 formulation start to decompose creating a large increase in fine particle emissions (Perricone et al.,
1096 2018). This transition temperature is affected by resin choice, with higher resin molecular weight

1097 (Joo et al., 2020) and higher heat resistance of binders (Joo et al., 2019) correlating well with low
1098 particle emission rates, particularly under higher temperature testing.

1099

1100 The location at which the emissions are formed is that of the friction contact. The dissipation of
1101 energy at this contact results in locally high temperatures, particularly on the pads (which explains
1102 the impact of fibre choice on particle emissions).

1103

1104 The temperature achieved, which relates to the particulate emissions is influenced by the thermal
1105 behaviour of both the brake pad and disc. Thermal conductivity of the brake disc has consequently
1106 been shown to influence the particle emissions, with particle emissions increasing as disc thermal
1107 conductivity reduces, even when the lower thermal conductivity is only at the surface through a thin
1108 oxynitride layer (Seo et al., 2021). Hardening of the disc has been demonstrated to reduce particle
1109 emissions as a result of the harder material behaviour as part of the friction surface (Perricone et al.,
1110 2018).

1111

1112 **7.3 Vehicle Design Decisions**

1113 The importance of brake material temperatures highlights the influence of vehicle design decisions
1114 on brake particle emissions. Frictional power has been shown to be a significant metric determining
1115 the particle emission rates (Alemani et al., 2018), disproportionately increasing emissions when
1116 considered on a per unit energy dissipated basis likely as a combined effect of contact temperatures
1117 and contact area. This suggests that vehicle mass and regenerative braking systems, both of which
1118 reduce the frictional power demand for a given deceleration event, will significantly influence
1119 particle emission rates. Although regenerative braking will reduce frictional power demand, there is
1120 a trend of increasing vehicle weight (EPA, 2021) which will somewhat counter this. Vehicle
1121 aerodynamic design, specifically with respect to brake cooling and airflows will also affect brake
1122 contact temperatures and therefore emissions.

1123 Studies which have compared enclosed and open brake systems (i.e. drum brakes versus disc
1124 brakes) have shown that the emitted airborne particulates can be ~10 times lower when the brake
1125 system is enclosed (Hagino et al., 2016). Drum brakes seem to have fallen out of favour in many
1126 markets, however, the drop in particle emissions is somewhat attributed to the enclosing of the
1127 friction surfaces which may give a technological opportunity to reduce particle emissions not
1128 dissimilar in some ways to a passive particle filter system.

1129

1130 **7.4 Brake Particle Filters/Traps**

1131 Filtration systems have been developed for many industrial applications and are now common in
1132 automotive internal combustion engine exhaust systems. This technology is translating across to
1133 brake systems with examples of forced induction filtration systems (developed by Tallano) (Hascoet
1134 and Adamczak, 2020; Gramstat et al., 2020) and passive filtration systems (developed by Mann
1135 Hummel) (Bock et al., 2019).

1136

1137 The Tallano system uses a grooved pad with an extraction system drawing air out of the groove
1138 through a filter using an electric motor powered from the vehicle electrical system. The system has
1139 been tested on brake dynamometer rigs and on vehicle. Mass based measurement was primarily
1140 used to determine the trapping effectiveness by comparing the weight loss of the friction materials
1141 to the weight gain of the filter, supported by particle measurements. The results indicate that
1142 filtration efficiencies between 40 and 90% are achievable depending on the braking behaviour,
1143 particle size range and filter condition (Hascoet and Adamczak, 2020). The filtration efficiency
1144 appeared to degrade to the lower levels when brake temperatures exceeded the decomposition
1145 temperatures – suggesting potential difficulties in this particular design in trapping finer particles.
1146 Comments on filter capacity suggest that limited life of the filter (consumable) may still need to be
1147 resolved as well as understanding its behaviour as a function of weather (Gramstat et al., 2020).

1148

1149 The Mann-Hummel system consists of a passive filter element located downstream of the brake
1150 pads and caliper with the intention of using the induced flows to transport the airborne particles
1151 through a filter layer. This has the advantage of being retrofittable due to the minimal changes to the
1152 existing braking system (Bock et al., 2019). It is reported to have been demonstrated on many trial
1153 vehicles under many conditions but limited supporting data is currently published. They claim filter
1154 life comparable with pad life and filtration efficiencies as high as 80%. The flow rate through the
1155 filter will depend on the filter permeability which will change over time, thereby changing the
1156 bypass ratios and filtration efficiencies. Conceptually similar filters have reported unevidenced
1157 filtration efficiencies of up to 92% (Fieldhouse and Gelb, 2016). Partial flow filters used in exhaust
1158 systems exhibit similar behaviour with filter loading and flow condition, achieving filtration
1159 efficiencies between 30 and 70% (Schrewe et al., 2012). This comparison should clearly be taken
1160 with caution, but does indicate the degree of variation that is typical of partial flow systems.

1161

1162 **7.5 Road Surface Cleaning**

1163 Road maintenance has been implemented for many generations to reduce the debris and loose
1164 matter on the streets. Technologies to achieve this include sweeping systems, vacuum systems
1165 (often combined with sweeping) and street washing.

1166

1167 There have been a number of studies attempting to elucidate the effect of street sweeping,
1168 vacuuming and washing on airborne particulate number, the majority of which have struggled to
1169 identify a statistically significant effect due to the influence of, for example meteorological effects.
1170 Daily averages were highlighted to be a problematic metric for identifying effects of road
1171 maintenance. The majority of the older studies have already been reviewed well (Amato et al.,
1172 2010b) to which the reader is directed. The difficulties in measuring PM change amongst other
1173 external effects meant that many studies were inconclusive. The minority identified improvements
1174 in PM₁₀ of the order of 2-3 $\mu\text{g}\cdot\text{m}^{-3}$ and 6-30% reduction. The most significant differences arose in

1175 the hours immediately after the road maintenance, with almost twice the reduction in coarse
1176 particles compared to fine particles (Amato et al., 2010a).

1177

1178 Since that review (Amato et al., 2010b), there have been few focused studies on the effects of street
1179 maintenance on airborne PM. A longer duration study in Nevada US, overcame many of the issues
1180 with short term studies through analysis of a year's worth of data alongside street maintenance data
1181 (Zhu et al., 2012). They concluded that winter road treatments have an extended impact on
1182 emissions throughout subsequent months through the transport and degradation of traction additives
1183 (e.g. sand, gravel and salt), with the recommendation to perform street maintenance (sweeping and
1184 vacuuming) after winter storms to remove the remaining traction materials having recognised
1185 statistically significant improvements in roads with similar maintenance schedules. Other analysis
1186 focused studies indicate benefits of $\sim 2 \mu\text{g}\cdot\text{m}^{-3}$ and 2-15% reduction in PM_{10} (Karanasiou et al.,
1187 2011).

1188

1189 The importance of transport of emission material from secondary and tertiary roads was highlighted
1190 by Zhu et al. (2012) with similar results in a more recent study reported by Monks et al. (2019). In
1191 this latter 2017 study a road which was heavily loaded with suspendable matter due to industrial use
1192 was subject to street maintenance. Combined sweeping and washing contributed to a measurable
1193 benefit in PM_{10} whereas sweeping alone was reported not to, supporting earlier conclusions from
1194 Amato et al. (2010b).

1195

1196 **7.6 Dust Suppressants**

1197 For unpaved roads (particularly those in mining industries), dust suppressants are being widely used
1198 to help bind the surface road matter and reduce the amount of transfer to airborne PM. Water can
1199 have a short-term benefit widely recognised through meteorological effects in other studies and

1200 industrial applications of water sprays, however, dust suppressants aim to have a longer lasting
1201 effect. Typical formulations include salts, polymers and resins.

1202

1203 Results, like those for street sweeping, vary depending on the location, with focused reviews on this
1204 area showing benefits in areas of high surface loading of suspendable materials (e.g. areas with
1205 significant use of winter road traction additives, industrial areas or unpaved roads), with benefits of
1206 typically 25-50% in PM₁₀ (Memorandum of Understanding, 2015). The effect of dust suppressants
1207 on industrial roads in Spain identified an ~8% reduction in kerbside PM₁₀ compared to ~18% for
1208 street washing (Amato et al., 2016d). There is a distinct lack of available recent literature for more
1209 lightly loaded road surfaces.

1210

1211 Research communities, particularly for the mining industries continue to develop dust suppressants
1212 with improved durability and effectiveness through improved mechanical properties and ability to
1213 be robust to wetting events, for example through the careful use of hydrophilic and hydrophobic
1214 components in a polymer dust suppressant to improve water durability (Lo et al., 2020). However,
1215 the application is focused on centimetres of depth of loose material and therefore offers little insight
1216 into the effects on paved roads where the resuspendable layer is substantially thinner.

1217

1218 **8. BATTERY ELECTRIC AND AUTONOMOUS VEHICLES**

1219 Many countries have now set deadline dates for terminating the sale of light duty internal
1220 combustion engine vehicles (ICEVs), after which new passenger car sales are expected to be
1221 restricted to battery electric vehicles. The average battery electric vehicle is heavier than the nearest
1222 equivalent ICEV, and Timmers and Achten (2016) raised the possibility that non-exhaust particle
1223 emissions from such a vehicle would exceed all emissions, including exhaust, from an ICEV. Their
1224 estimates were based upon assumed increases in tyre and road surface wear, and greater
1225 resuspension due to increased vehicle mass, and included much reduced brake wear due to

1226 regenerative braking (a system in which vehicle inertia is used to rotate the electric motor, which
1227 then acts as a generator) on electric vehicles. A scenario in which battery electric vehicles became
1228 mandatory in Ireland (Alam et al., 2018) showed an increase in PM_{2.5} emissions based upon the
1229 emission factors assumed by Timmers and Achten (2016).

1230

1231 Beddows and Harrison (2021) used the vehicle weight dependence of emission factors to estimate
1232 the consequences of a change from an ICEV to a battery electric passenger car. The outcomes
1233 differed for gasoline and diesel vehicles, primarily due to a weight difference, and between rural,
1234 urban and motorway driving due to differing levels of wear emissions according to driving mode.
1235 However, the largest influence was the reduction in brake wear due to regenerative braking. A
1236 review by OECD (2020) found estimates of the reduction in brake wear ranging from 25-95%.
1237 Based upon an estimate of 90% reduction due to regenerative braking, Beddows and Harrison
1238 (2021) report reductions of up to 23% in total emissions of PM_{2.5} (including exhaust for the ICEV)
1239 for a battery electric vehicle during urban driving, with smaller improvements in rural and
1240 motorway driving. OECD (2020) used similar methods of calculation, but assumed identical
1241 emissions from gasoline and diesel vehicles, and a 75% reduction in brake wear due to regenerative
1242 braking. They calculated non-exhaust emission factors for passenger cars, SUV and light
1243 commercial vehicles with ranges of 100 and 300km, finding net reductions in PM_{2.5} emissions of
1244 11.2 – 13.3% for the 100 km range vehicles and an increase of 2.6 – 7.8% for the 300 km range
1245 vehicles. Exhaust emissions were not however included. Both studies therefore show little impact
1246 of electrification of the vehicle fleet upon PM_{2.5} emissions, with broadly similar conclusions for
1247 emissions of PM₁₀.

1248

1249 There are, as yet few hard data regarding autonomous (self-driving) vehicles. Current opinion is
1250 that the additional weight of batteries required for the computerisation of operation would
1251 detrimentally affect emissions compared to a conventional BEV, while the reduced braking due to

1252 smoother operation would be beneficial. The balance between the two has yet to be tested in
1253 practice.

1254

1255 **9. NON-EXHAUST EMISSIONS OF VOLATILE ORGANIC COMPOUNDS**

1256 Whilst particulate emissions are the most frequently cited atmospheric emission arising from
1257 vehicle non-exhaust sources, and currently the largest by mass, some vehicles are associated with
1258 non-exhaust emissions of VOCs. VOC emissions from road transport have reduced dramatically
1259 over the last thirty years, both from the tailpipe and from the evaporative loss of fuels. Emissions
1260 from the tailpipe (all vehicle types) in the United Kingdom reduced from 594 kt pa in 1990 to ~16
1261 kt in 2017 (NAEI, 2020)¹, driven by the universal adoption of the highly effective three-way
1262 catalytic convertor. Tailpipe emissions of VOCs typically comprise a mixture of unburned fuel
1263 (most significantly, toluene, xylenes and a range of C4-C10 alkanes), and products from partial
1264 combustion conditions (such as ethene, formaldehyde, acetaldehyde, propene) (NAEI, 2020).

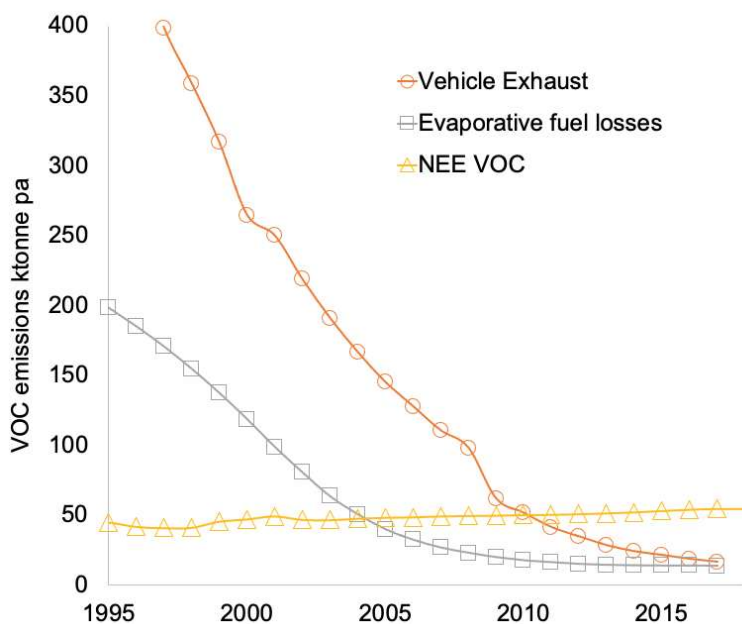
1265

1266 There are non-exhaust emissions of VOCs from ICE vehicles linked to the evaporative loss of fuel
1267 (predominantly gasoline vapour but also some emissions of intermediate volatility VOCs from
1268 diesel fuel). The speciation of those emissions is unsurprisingly linked to fuel formulation, in the
1269 UK the dominant VOCs released from evaporative fuel loss are C6 and C5 alkanes, xylenes,
1270 ethanol, butane, propane and toluene along with smaller amounts of a range of other alkanes and
1271 alkenes (NAEI, 2020). Technical controls on non-exhaust fuel evaporative losses have also been
1272 highly effective, reducing from 261 kt pa in the UK in 1990 to 13 kt in 2017. In combination, the
1273 emission from tailpipe and fuel loss can be considered as emissions that are intrinsic to ICE
1274 vehicles, and that over time are likely to be eliminated as these are replaced by EV/ low carbon
1275 equivalents.

¹The UK reports emissions from road transport separately on a 'fuel sold' and on a 'fuel used' basis. For tracking compliance with the UK's emissions ceilings, the 'fuel used' figures are used derived from kilometres driven. The UK figures shown in this report are on a fuel used basis.

1276 An often-overlooked component of non-exhaust VOC emissions are those associated with solvents
1277 used in ancillary systems and for car care. Most notable are screen-wash and windscreen de-icer.
1278 These emissions are agnostic to the propulsion system and will likely persist even with low carbon
1279 vehicle fleets. The non-exhaust emissions from this class fall into two types, non-aerosol and
1280 aerosol (referred to in EMEP reporting for inventories as: Non-aerosol product – automotive
1281 products NFR Code 2D3a 258 and Aerosols - car care NFR Code 2D3a 274). In combination, these
1282 non-exhaust VOC emissions from vehicles are estimated in the UK at ~55 kt pa, around 3.4 times
1283 the amount of VOC emitted from vehicle tailpipes in that year. Considered in the broader context,
1284 non-exhaust emissions of VOCs (and excluding fuel evaporation) represents around 6.8% of total
1285 UK anthropogenic VOC emissions. Non-aerosol non-exhaust emissions from screen-wash are
1286 dominated by VOCs such as ethanol, ethylene glycol, methanol and 2-methoxy ethanol. Methanol
1287 is particularly notable because of its direct toxicity and EU regulations introduced using the
1288 REACH Directive in 2018 limited its use in screen-wash to 0.6% v/v, now replaced most
1289 commonly by ethanol. Use of aerosol care car products (predominantly for de-icer, but other
1290 sources such as cleaners and polishes contribute also) releases *n*- and *iso*-butane and propane (as the
1291 propellant) and ethanol / 2 propanol as the active ingredient for de-icing. The trends since 1995 in
1292 the UK exhaust and fuel evaporative and NEE VOCs associated with all road transport for the UK
1293 are show in Figure 3.

1294



1295

1296

1297 **Figure 3:** UK emissions of NMVOCs from all road transport by sources according to the United
 1298 Kingdom National Atmospheric Emissions Inventory, summary data available at
 1299 <https://naei.beis.gov.uk>. Methodology for the construction of the VOC element of the NAEI is
 1300 described in Passant et al., 2002.

1301

1302 Whilst the total mass of emissions of VOC from non-exhaust sources is now dominated by ancillary
 1303 emissions from care car products, the differences in speciation between the various non-exhaust
 1304 sources are worth noting. Considering the Photochemical Ozone Creation Potentials (POCP) of the
 1305 two different NEE VOC sources, evaporative fuels are a more significant ozone source on a tonne-
 1306 for-tonne emission basis, since these contain some highly OH-reactive species such as
 1307 monoaromatics and alkenes. The alcohols which make up a significant component of VOCs from
 1308 aerosols and de-icers, whilst still capable of forming ozone, are somewhat lower reactivity. The
 1309 exact ozone formation potentials from different NEE VOC mixtures will depend on the wider
 1310 photochemical environment and is beyond this review.

1311

1312 In summary, whilst the total mass of VOCs emitted from vehicles has reduced dramatically over the
 1313 last three decades, there remains a significant non-exhaust source of emissions, through the release
 1314 of solvents used as consumable products in screen-wash and windscreen de-icing. NEE solvent

1315 emissions from vehicles substantially exceed the total currently being emitted from the tailpipe and
1316 from evaporative fuel losses and would likely persist with future decarbonised fleets.

1317

1318 **10. FUTURE TRENDS**

1319 With traffic levels predicted to grow in many countries, non-exhaust emissions of PM are expected
1320 to further increase if no actions are taken to control them. The UK's emissions inventory predicts
1321 emissions from tyre and brake wear and road abrasion will increase by 13% by 2030, relative to
1322 2018 levels. With exhaust emissions, and emissions from other sources, expected to decline over
1323 this time period as new vehicles meeting tighter exhaust emission standards enter the fleet, non-
1324 exhaust emissions are predicted to be responsible for 90% of all road transport emissions and 10%
1325 of all UK primary emissions of PM_{2.5} by 2030. Emission projections for all countries reported in
1326 this section are taken from official inventory reporting to EMEP Centre on Emission Inventories
1327 and Projections (EMEP, 2021)

1328

1329 The future trend is expected to be similar elsewhere in the world where traffic is predicted to
1330 increase. Emission inventory projections reported by countries in Europe for PM_{2.5}, all excluding
1331 resuspension, are not always complete or comparable making it difficult to compare predicted
1332 trends in non-exhaust emissions between countries. Many countries do predict a growth in non-
1333 exhaust emissions or for the trend to remain roughly flat, although some countries do consider a
1334 reduction in emissions will occur without any clear explanation why. According to the latest
1335 available inventory data, France and Germany each predict a 2% and 4% increase in non-exhaust
1336 emissions of PM_{2.5} from 2018 to 2030, respectively, with the share in these emissions to overall
1337 road transport emissions increasing from 42% to 74% in France and from 62% to 86% in Germany,
1338 a trend similar to the UK situation (EMEP, 2021). Based on inventories for all those countries in
1339 Europe which have made national inventory predictions of non-exhaust emissions, the contribution
1340 made by these sources to total road transport emissions of PM_{2.5} increases, on average, from 50% in

1341 2018 to 71% by 2030. Therefore, the overall picture in Europe is clearly one in which non-exhaust
1342 emissions are expected to become the dominant source of overall traffic emissions of PM by 2030.

1343

1344 Data on future trends in emissions are sparse for other countries and regions of the world. The
1345 California Air Resources Board (CARB) has a comprehensive set of inventory and projections data
1346 showing a similar trend in non-exhaust emissions as in Europe (CARB, 2016; 2017). As described
1347 earlier, the California inventory includes emissions from tyre and brake wear and resuspension and
1348 these are predicted to increase by 11% and 12%, respectively, from 2018 to 2030. To compare with
1349 the scope of PM_{2.5} emissions shown in Table 1 for 2018, tyre and brake wear emissions combined
1350 as a share of total road transport emissions in California (i.e. excluding resuspension) are expected
1351 to increase from 71% in 2018 to 89% in 2030. Including resuspension further increases the
1352 contribution of non-exhaust emission in future years.

1353

1354 Future trends in non-exhaust emissions according to emission inventories are largely based on
1355 predictions in traffic growth, changes in the vehicle fleet and emission factors, but the general
1356 assumption is made that emission factors for non-exhaust sources on a g/km basis remain
1357 unchanged. A major uncertainty in these predictions is whether this is the case and whether PM
1358 factors will change with changes in vehicle design, weight and tyre and brake wear materials and
1359 technologies. A particular area of uncertainty is the effect the future fleet penetration of hybrid and
1360 battery electric vehicles will have on non-exhaust emissions. As discussed in previous chapters,
1361 emissions may be impacted in either direction through the effect of increased vehicle mass which
1362 may increase emissions and the effect of regenerative braking which may reduce emissions. There
1363 is an urgent need for measurements to confirm emission factors for these vehicle types as there are
1364 currently no confirmed or widely accepted emission factors for use in inventories that reflect the
1365 effect these technologies have on tyre and brake wear emissions under real-world conditions.

1366 The report by the OECD on non-exhaust emissions estimated that there would be a 53.5% increase
1367 in non-exhaust emissions of PM from passenger vehicles worldwide, from 0.85 million tonnes in
1368 2017 to 1.3 million tonnes in 2030 based on the predicted growth in the global vehicle stock
1369 (OECD, 2020). This was for a Business-as-Usual scenario that assumed an uptake of lightweight
1370 electric vehicles (EVs) to 4% of the vehicle stock by 2030 and the emission estimates considered
1371 the effect of vehicle weight and regenerative braking on emissions. The study concluded that a
1372 doubling of the EV numbers to 8% of the vehicle stock would have a minimal effect on the rate of
1373 increase in non-exhaust emissions. However, higher emissions would occur if the uptake were with
1374 heavier, larger range EVs.

1375

1376 Beddows and Harrison (2021) recently considered the dependence of non-exhaust emissions from
1377 battery electric vehicles on vehicle mass. The study considered the issue on a g/km emission factor
1378 basis for tyre and brake wear, road abrasion and resuspension under different driving conditions. It
1379 concluded that overall there would only be a modest change in total PM emissions from a passenger
1380 car built to current Euro 6 standards as gains made by removal of the exhaust emissions are offset
1381 by the likely increase in non-exhaust emissions due to the increased weight of EVs. However, the
1382 outcome is critically dependent upon the extent of regenerative braking which is essential if there is
1383 to be a net benefit of EVs relative to conventional internal combustion engine vehicles. Studies like
1384 this highlight the importance of gathering relevant emission factors for each aspect of non-exhaust
1385 emissions from EVs if accurate predictions are to be made on future trends in PM emissions from
1386 the global vehicle fleet as increasing numbers of EVs enter service.

1387

1388 **11. CONCLUSIONS**

1389 Nearly all non-exhaust vehicle emissions (brake wear, tyre wear, road wear and resuspension
1390 particles, and vehicle-care VOCs) arise regardless of the type of vehicle and its mode of power. No
1391 legislation is currently in place specifically to target non-exhaust emissions, in contrast to

1392 progressively stringent legislation that has been effective in reducing exhaust emissions.
1393 Consequently, in most developed countries national emissions inventories indicate that non-exhaust
1394 particle and VOC emissions now exceed their respective exhaust emissions. Future projections are
1395 for non-exhaust emissions to be an increasingly dominant proportion of road transport-related
1396 particle and VOC emissions. From an air quality perspective it is therefore essential to recognise
1397 that further reductions in PM_{2.5} and PM₁₀ in relation to road sources requires attention to non-
1398 exhaust as well as to exhaust emissions.

1399

1400 Comparison between inventories for non-exhaust particle emissions in different regions of the
1401 world is complicated by the different scope in the sources that are covered, for instance with
1402 resuspension being included in inventories reported in the U.S. and Latin America, but not in
1403 Europe. Nevertheless, there is a consistent trend showing at least 50% of PM_{2.5} emissions from
1404 road transport come from tyre, brake wear and road abrasion in much of Europe and the U.S. while
1405 estimates of resuspension in the U.S. suggest around a further 50% occur from this source meaning
1406 a total dominance in all non-exhaust source emissions.

1407

1408 Non-exhaust emission factors used in inventories are sparse, have not been updated since the 1990s
1409 and span a wide range. Therefore, the exact contribution of non-exhaust emissions to air quality
1410 locally and nationally is subject to considerable uncertainty. It is strongly recommended that efforts
1411 continue on development of consistent approaches internationally for measurement of non-exhaust
1412 emission factors. Methods must strive to capture real-world variation in non-exhaust emissions
1413 arising from different materials used, type of road, surface wetness, different cycles of braking,
1414 cornering, etc.

1415

1416 Measurement evidence is clear that non-exhaust emissions increase concentrations of PM₁₀ and
1417 PM_{2.5} and some metals at roadside. Studies suggest contributions (including resuspension) of

1418 several (4-5) $\mu\text{g m}^{-3}$ of PM_{10} at busy roadsides, mostly in $\text{PM}_{2.5-10}$, and in the region 1-2 $\mu\text{g m}^{-3}$ for
1419 the urban background in central London.

1420

1421 Emission inventories for non-exhaust PM sources are largely based on old measurements and do not
1422 account for contemporary vehicle designs, technologies, tyre and brake systems and materials.

1423 Emission factors are held constant at current values and assumed not to change over time. The
1424 effect of electrification of the vehicle fleet on non-exhaust PM emissions is highly uncertain.

1425 Important future determinants of non-exhaust particle emissions are the extent of regenerative
1426 braking in the vehicle fleet and vehicle mass. The former eliminates direct brake-wear emissions
1427 but where batteries, sensors and computer hardware increases vehicle mass, the result may be
1428 greater tyre, road and resuspension emissions. A limited number of recent studies show little impact
1429 of electrification of the vehicle fleet upon $\text{PM}_{2.5}$ emissions, with broadly similar conclusions for
1430 emissions of PM_{10} .

1431

1432 Other non-exhaust emission mitigation approaches (in addition to driving less) include: managing
1433 driving patterns towards lower speeds and less braking; on-vehicle brake-wear capture;
1434 development of low-wear tyres and road surfaces; and road sweeping/washing and application of
1435 dust suppressants to road surfaces, although there is little evidence at present that the latter
1436 measures are effective at mitigating non-exhaust particle emissions in the long term.

1437

1438 Technological innovation on reducing non-exhaust emissions is encouraged, including via
1439 regulation, but the net effect of all on-vehicle changes (e.g. regenerative braking, particle capture,
1440 autonomous driving, vehicle mass) must be evaluated holistically across all sources of non-exhaust
1441 emissions, including any trade-offs between particle mass and particle number emissions.

1442

1443

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1446 the inventory data for NEE.

1447

1448 **CONFLICT OF INTERESTS**

1449 The authors declare no competing financial interest.

1450

1451 **REFERENCES**

1452

1453 Aatmeeyata, Sharma, M., 2010. Polycyclic aromatic hydrocarbons, elemental and organic carbon
1454 emissions from tire-wear, *Sci. Tot. Environ.*, 408, 4563-4568.

1455

1456 Adachi, K., Tainosho, Y., 2004. Characterization of heavy metal particles embedded in tire dust,
1457 *Environ. Int.*, 30, 1009-1017.

1458

1459 Adamiec, E., Jarosz-Krzeminska, E., Wieszala, R., 2016. Heavy metals from non-exhaust vehicle
1460 emissions in urban and motorway road dusts, *Environ. Monitor. Assess.*, 188, 369, doi
1461 10.1007/s10661-016-5377-1.

1462

1463 Alam Md.S., Hyde, B., Duffy, P., McNabola, A., 2018. Analysing the Co-Benefits of transport
1464 fleet and fuel policies in reducing PM_{2.5} and CO₂ emissions, *J. Clean. Prod.*, 172, 623-634.

1465

1466 Alemani, M., Wahlstrom, J., Olofsson, U., 2018. On the influence of car brake system parameters
1467 on particulate matter emissions, *Wear*, 396-397, 67-74.

1468

1469 Alves, C.A., Vicente, E.D., Vicente, A.M.P., Rienda, I.C., Tomé, M., Querol, X., Amato, F., 2020.
1470 Loadings, chemical patterns and risks of inhalable road dust particles in an Atlantic city in the north
1471 of Portugal, *Sci. Tot. Environ.*, 737, 139596.

1472

1473 Alves, C.A., Evtyugina, M., Vicente, A.M.P., Vicente, E.D., Nunes, T.V., Silva, P.M.A., Duarte,
1474 M.A.C., Pio, C.A., Amato, F., Querol, X., 2018. Chemical profiling of PM₁₀ from urban road dust,
1475 *Sci. Total Environ.*, 634, 41-51.

1476

1477 Alves, C.A., Evtyugina, M., Vicente, A., Conca, E., Amato, F., 2021. Organic profiles of brake
1478 wear particles, *Atmos. Res.*, 255, 105557

1479

1480 Amato, F., Alastuey, A., Karanasiou, A., Lucarelli, F., Nava, S., Calzolari, G., Severi, M., Becagli,
1481 S., Gianelle, V. L., Colombi, C., Alves, C., Custodio, D., Nunes, T., Cerqueira, M., Pio, C.,
1482 Eleftheriadis, K., Diapouli, E., Reche, C., Minguillón, M. C., Manousakas, M. I., Maggos, T.,
1483 Vratolis, S., Harrison, R.M., and Querol, X., 2016a. AIRUSE-LIFE+: a harmonized PM speciation
1484 and source apportionment in five southern European cities, *Atmos. Chem. Phys.*, 16, 3289-3309.

1485

1486 Amato, F., Favez, O., Pandolfi, M., Alastuey, A., Querol, X., Moukhtar, S., Bruge, B., Verlhac, S.,
1487 Orza, J.A.G., Bonnaire, N., Le Priol, T., Petit, J. F., Sciare, J., 2016b. Traffic induced particle
1488 resuspension in Paris: Emission factors and source contributions, *Atmos. Environ.*, 129, 114-124.

1489

1490 Amato, F., Zandveld, P., Keuken, M., Jonkers, S., Querol, X., Reche, C., Denier van der Gon,
1491 H.A.C., Schaap, M., 2016c. Improving the modeling of road dust levels for Barcelona at urban scale
1492 and street level, *Atmos. Environ.*, 125, 231-242.

1493

1494 Amato, F., Escrig, A., Sanfelix, V., Celades, I., Reche, C., Monfort, E., Querol, X., 2016d. Effects
1495 of water and CMA in mitigating industrial road dust resuspension, *Atmos. Environ.*, 131, 334-340.

1496

1497 Amato, F., Cassee, F.R., Denier van der Gon, H.A.C., Gehrig, R., Gustafsson, M., Hafner, W.,
1498 Harrison, R.M., Jozwicka, M., Kelly, F.J., Moreno, T., Prevot, A.S.H., Schaap, M., Sunyer, J.,
1499 Querol, X., 2014. Urban air quality: The challenge of traffic non-exhaust emissions, *J. Hazard.
1500 Mater.*, 275, 31-36.

1501

1502 Amato, F., Karanasiou, A., Moreno, T., Alastuey, A., Orza, J. A. G., Lumbreras, J., Borge, R.,
1503 Boldo, E., Linares, C., Querol, X., 2012a. Emission factors from road dust resuspension in a
1504 Mediterranean freeway, *Atmos. Environ.*, 61, 580-587.
1505
1506 Amato, F., Schaap, M., Denier van der Gon, H.A.C., Pandolfi, M., Alastuey, A., Keuken, M.,
1507 Querol, X., 2012b. Effect of rain events on the mobility of road dust load in two Dutch and Spanish
1508 roads, *Atmos. Environ.*, 62, 352-358.
1509
1510 Amato, F., Hopke, P. K., 2012. Source apportionment of the ambient PM_{2.5} across St. Louis using
1511 constrained positive matrix factorization, *Atmos. Environ.*, 46, 329-337.
1512
1513 Amato, F., Pandolfi, M., Moreno, T., Furger, M., Pey, J., Alastuey, A., Bukowiecki, N., Prevot, A.
1514 S. H., Baltensperger, U., Querol, X., 2011. Sources and variability of inhalable road dust particles
1515 in three European cities, *Atmos. Environ.*, 45, 6777-6787.
1516
1517 Amato, F., Nava, S., Lucarelli, F., Querol, X., Alastuey, A., Baldasano, J. M., Pandolfi, M., 2010a.
1518 A comprehensive assessment of PM emissions from paved roads: Real-world Emission factors and
1519 intense street cleaning trials, *Sci. Tot. Environ.*, 408, 4309-4318.
1520
1521 Amato, F., Querol, X., Johansson, C., Nagl, C., Alastuey, A., 2010b. A review on the effectiveness
1522 of street sweeping, washing and dust suppressants as urban PM control methods, *Sci. Tot. Environ.*,
1523 408, 3070-3084.
1524
1525 Amato, F., Pandolfi, M., Viana, M., Querol, X., Alastuey, A., Moreno, T., 2009a. Spatial and
1526 chemical patterns of PM₁₀ in road dust deposited in urban environment, *Atmos. Environ.*, 43, 1650-
1527 1659.
1528
1529 Amato, F., Pandolfi, M., Escrig, A., Querol, X., Alastuey, A., Pey, J., Perez, N., Hopke, P. K.,
1530 2009b. Quantifying road dust resuspension in urban environment by Multilinear Engine: A
1531 comparison with PMF2, *Atmos. Environ.*, 43, 2770-2780.
1532
1533 AQEG, 2019. Non-Exhaust Emissions from Road Traffic, report of the Air Quality Expert Group
1534 to the UK Department for Environment, Food and Rural Affairs; Scottish Government; Welsh
1535 Government and Department of the Environment in Northern Ireland, [https://uk-
1536 air.defra.gov.uk/assets/documents/reports/cat09/1907101151_20190709_Non_Exhaust_Emissions
1537 typeset_Final.pdf](https://uk-air.defra.gov.uk/assets/documents/reports/cat09/1907101151_20190709_Non_Exhaust_Emissions_typeset_Final.pdf).
1538
1539 Archer-Nicholls, S., Lowe, D., Utembe, S., Allan, J., Zaveri, R.A., Fast, J.D., Hodnebrog, Ø.,
1540 Denier van der Gon, H., McFiggans, G., 2014, Gaseous chemistry and aerosol mechanism
1541 developments for version 3.5.1 of the online regional model, WRF-Chem, *Geosci. Model Develop.*,
1542 7, 2557-2579.
1543
1544 Avagyan, R., Sadiktsis, I., Bergvall, C., Westerholm, R., 2014. Tire tread wear particles in ambient
1545 air—a previously unknown source of human exposure to the biocide 2-mercaptobenzothiazole,
1546 *Environ. Sci. Pollut. Res. Int.*, 21, 11580-11586.
1547
1548 Baensch-Baltruschat, B., Kocher, B., Stock, F., Reifferscheid, G., 2020. Tyre and road wear
1549 particles (TRWP) - A review of generation, properties, emissions, human health risk, ecotoxicity,
1550 and fate in the environment, *Sci. Tot. Environ.*, 733, 137823.
1551

1552 Beddows, D.C.S., Harrison, R.M., 2021. PM₁₀ and PM_{2.5} emission factors for non-exhaust
1553 particles from road vehicles: Dependence upon vehicle mass and implications for battery electric
1554 vehicles, *Atmos. Environ.*, 244, 117886.
1555
1556 Beddows, D.C.S., Dall'Osto, M., Olatunbosun, O.A., Harrison, R.M., 2016. Detection of brake
1557 wear aerosols by aerosol time-of-flight mass spectrometry, *Atmos. Environ.*, 129, 167-175.
1558
1559 Beddows, D.C.S., Harrison, R.M., Green, D.C., Fuller, G.W., 2015. Receptor modelling of both
1560 particle composition and size distribution from a background site in London, UK, *Atmos. Chem.
1561 Phys.*, 15, 10107-10125.
1562
1563 Blau, P.J., 2001. Compositions, Functions, and Testing of Friction Brake Materials and Their
1564 Additives, Report No. ORNL/TM-2001/64, United States, doi:10.2172/788356.
1565
1566 Blok, J., 2005. Environmental exposure of road borders to zinc, *Sci. Tot. Environ.*, 348, 173-190.
1567
1568 Bock, L., Kolczyk, M., Drummond, R., Klein, G-M., Lehmann, M., Jessberger, T., Kümmerling,
1569 V., Thébault, E., Beck, A., 2019. The Brake Dust Particle Filter for Fine Dust Reduction, EBS-014,
1570 EuroBrake 2019, 21-23 May, International Congress Center, Dresden, Germany,
1571 <http://2019.eurobrake.net/>.
1572
1573 Boulter, P.G., 2005. A Review of Emission Factors and Models for Road Vehicle Non-Exhaust
1574 Particulate Matter, Report No. PPR065 prepared by TRL Limited for Defra [https://uk-](https://uk-air.defra.gov.uk/library/reports?report_id=364)
1575 [air.defra.gov.uk/library/reports?report_id=364](https://uk-air.defra.gov.uk/library/reports?report_id=364).
1576
1577 Bukowiecki, N., Lienemann, P., Hill, M., Furger, M., Richard, A., Amato, F., Prévôt, A.S.H.,
1578 Baltensperger, U., Buchmann, B., Gehrig, R., 2010. PM₁₀ emission factors for non-exhaust
1579 particles generated by road traffic in an urban street canyon and along a freeway in Switzerland,
1580 *Atmos. Environ.*, 44, 2330-2340.
1581
1582 Bukowiecki, N., Lienemann, P., Hill, M., Figi, R., Richard, A., Furger, M., Rickers, K., Falkenberg,
1583 G., Zhao, Y., Cliff, S.S., Prevot, A.S.H., Baltensperger, U., Buchmann, B., Gehrig, R., 2009. Real-
1584 world emission factors for antimony and other brake wear related trace elements: size-segregated
1585 values for light and heavy duty vehicles, *Environ. Sci. Technol.*, 43, 8072-8078.
1586
1587 Bukowiecki, N., Hill, M., Gehrig, R., Zwicky, C.N., Lienemann, P., Hegedüs, F., Falkenberg, G.,
1588 Weingartner, E., Baltensperger, U., 2005. Trace Metals in Ambient Air: Hourly Size-Segregated
1589 Mass Concentrations Determined by Synchrotron-XRF, *Environ. Sci. Technol.*, 39, 5754-5762.
1590
1591 CARB, 2017. California Air Resources Board EMFAC2017 Web Database,
1592 <https://arb.ca.gov/emfac/2017/>.
1593
1594 CARB, 2016. CEPAM: 2016. SIP - Standard Emission Tool, California Air Resources Board,
1595 <https://ww2.arb.ca.gov/criteria-pollutant-emission-inventory-data>.
1596
1597 Carslaw, D.C., Taylor, P.J., 2009. Analysis of air pollution data at a mixed source location using
1598 boosted regression trees, *Atmos. Environ.*, 43, 3563-3570.
1599
1600 Chan, D., Stachowiak, G.W., 2004. Review of automotive brake friction materials, *Proc. Inst.
1601 Mech. Eng. Part D-J. Automob. Eng.*, 218, 953-966, 10.1243/0954407041856773.
1602

1603 Charron, A., Polo-Rehn, L., Besombes, J.L., Golly, B., Buisson, C., Chanut, H., Marchand, N.,
1604 Guillaud, G., Jaffrezo, J.L., 2019. Identification and quantification of particulate tracers of
1605 exhaust and non-exhaust vehicle emissions, *Atmos. Chem. Phys.*, 19, 5187-5207.
1606

1607 Chow, J., Watson, J., 2013. Chemical Analyses of Particle Filter Deposits, in: *Aerosols Handbook:
1608 Measurement, Dosimetry, and Health Effects*, 2nd Edition, Ruzer, L.S., Harley, N.H. (Eds), Taylor
1609 & Francis, Boca Raton.
1610

1611 Chow, J.C., Watson, J.G., 2002. Review of PM_{2.5} and PM₁₀ apportionment for fossil fuel
1612 combustion and other sources by the chemical mass balance receptor model, *Energ. Fuel*, 16, 222-
1613 260.
1614

1615 Chow, J.C., 1995. Measurement methods to determine compliance with ambient air-quality
1616 standards for suspended particles, *JAWMA*, 45, 320-382.
1617

1618 Councill, T.B., Duckenfield, K.U., Landa, E.R., Callender, E., 2004. Tire-wear particles as a
1619 source of zinc to the environment, *Environ. Sci. Technol.*, 38, 4206-4214.
1620

1621 Cowan, N., Blair, D., Malcolm, H., Graham, M., 2021. A survey of heavy metal contents of rural
1622 and urban roadside dusts: comparisons at low, medium and high traffic sites in Central Scotland,
1623 *Environ. Sci. Pollut. Res.*, 28, 7365-7378.
1624

1625 Crilley, L.R., Lucarelli, F., Bloss, W.J., Harrison, R.M., Beddows, D.C., Calzolari, G., Nava, S.,
1626 Valli, G., Bernardoni, V., Vecchi, R., 2017. Source apportionment of fine and coarse particles at a
1627 roadside and urban background site in London during the 2012 summer ClearfLo campaign,
1628 *Environ. Pollut.*, 220, Part B, 766-778.
1629

1630 Daellenbach, K.R., Uzu, G., Jiang, J., Cassagnes, L.-E., Leni, Z., Vlachou, A., Stefenelli, G.,
1631 Canonaco, F., Weber, S., Segers, A., Kuenen, J.J.P., Schaap, M., Favez, O., Albinet, A., Aksoyoglu,
1632 S., Dommen, J., Baltensperger, U., Geiser, M., El Haddad, I., Jaffrezo, J.-L., Prévôt, A.S.H., 2020.
1633 Sources of particulate-matter air pollution and its oxidative potential in Europe, *Nature*, 587,
1634 <https://doi.org/10.1038/s41586-020-2902-8>.
1635

1636 Dahl, A., Gharibi, A., Swietlicki, E., Gudmundsson, A., Bohgard, M., Ljungman, A., Blomqvist,
1637 G., Gustafsson, M., 2006. Traffic-generated emissions of ultrafine particles from pavement–tire
1638 interface, *Atmos. Environ.*, 40, 1314–1323.
1639

1640 Dall'Osto, M., Beddows, D.C.S., Gietl, J.K., Olatunbosun, O.A., Yang, X., Harrison, R.M., 2014.
1641 Characteristics of tyre dust in polluted air: Studies by single particle mass spectrometry
1642 (ATOFMS), *Atmos. Environ.*, 94, 224-230.
1643

1644 Dall'Osto, M., Querol, X., Amato, F., Karanasiou, A., Lucarelli, F., Nava, S., Calzolari, G., and
1645 Chiari, M., 2013. Hourly elemental concentrations in PM_{2.5} aerosols sampled simultaneously at
1646 urban background and road site during SAPUSS – diurnal variations and PMF receptor modelling,
1647 *Atmos. Chem. Phys.*, 13, 4375-4392.
1648

1649 Davis, A.P., Shokouhian, M., Ni, S., 2001. Loading estimates of lead, copper, cadmium, and zinc
1650 in urban runoff from specific sources, *Chemosphere*, 44, 997-1009, 2001.
1651

1652 de la Paz, D., Borge, R., Vedrenne, M., Lumbreras, J., Amato, F., Karanasiou, A., Boldo, E.,
1653 Moreno, T., 2015. Implementation of road dust resuspension in air quality simulations of particulate
1654 matter in Madrid (Spain), *Front. Environ. Sci.*, 3, doi: 10.3389/fenvs.2015.00072.

1655 De Silva, S., Ball, A.S., Indrapala, D.V., Reichman, S.M., 2021. Review of the interactions
1656 between vehicular emitted potentially toxic elements, roadside soils, and associated biota,
1657 Chemosphere, 263, 128135, 2021.
1658

1659 Denby, B.R., Kupiainen, K.J., Gustafsson, M., 2018. Review of Road Dust Emissions, in: Non-
1660 Exhaust Emissions, Amato, F. (Ed), Ch. 9, Academic Press, 183-203.
1661

1662 Denby, B.R., Sundvor, I., Johansson, C., Pirjola, L., Ketzler, M., Norman, M., Kupiainen, K.,
1663 Gustafsson, M., Blomqvist, G., Omstedt, G., 2013. A coupled road dust and surface moisture model
1664 to predict non-exhaust road traffic induced particle emissions (NORTRIP). Part 1: Road dust
1665 loading and suspension modelling, Atmos. Environ., 77, 283-300.
1666

1667 Denier van der Gon, H.A.C., Gerlofs-Nijland, M.E., Gehrig, R., Gustafsson, M., Janssen, N.,
1668 Harrison, R.M., Hulskotte, J., Johansson, C., Jozwicka, M., Keuken, M., Krijgsheld, K.,
1669 Ntziachristos, L., Riediker, M., Cassee, F. R., 2013. The policy relevance of wear emissions from
1670 road transport, now and in the future—an international workshop report and consensus statement,
1671 JAWMA, 63, 136-149.

1672 Denier van der Gon, H.A.C., Hulskotte, J.H.J., Visschedijk, A J.H., Schaap, M., 2007. A revised
1673 estimate of copper emissions from road transport in UNECE-Europe and its impact on predicted
1674 copper concentrations, Atmos. Environ., 41, 8697-8710.
1675

1676 Dong, S., Ochoa Gonzalez, R., Harrison, R. M., Green, D., North, R., Fowler, G., Weiss, D., 2017.
1677 nIsotopic signatures suggest important contributions from recycled gasoline, road dust and non-
1678 exhaust traffic sources for copper, zinc and lead in PM₁₀ in London, United Kingdom, Atmos.
1679 Environ., 165, 88-98.
1680

1681 Dongarrà, G., Manno, E., Varrica, D., 2008. Possible markers of traffic-related emissions, Environ.
1682 Monitor. Assess., 154, 117-125.
1683

1684 EEA, 2021. National emissions reported to the Convention on Long-range Transboundary Air
1685 Pollution (LRTAP Convention), European Environment Agency, [https://www.eea.europa.eu/data-
1686 and-maps/data/national-emissions-reported-to-the-convention-on-long-range-transboundary-air-
1687 pollution-lrtap-convention-14](https://www.eea.europa.eu/data-and-maps/data/national-emissions-reported-to-the-convention-on-long-range-transboundary-air-pollution-lrtap-convention-14).
1688

1689 EMEP, 2021. EMEP Centre on Emission Inventories and Projections, [https://www.ceip.at/webdab-
1690 emission-database/reported-emissiondata](https://www.ceip.at/webdab-emission-database/reported-emissiondata).
1691

1692 EMEP/EEA, 2019. EMEP/EEA Air Pollutant Emission Inventory Guidebook 2019, Road
1693 Transport Tyre and Brake Wear and Road Abrasion, [https://www.eea.europa.eu/publications/emep-
1694 eea-guidebook-2019/part-b-sectoral-guidance-chapters/1-energy/1-a-combustion/1-a-3-b-vi/view](https://www.eea.europa.eu/publications/emep-eea-guidebook-2019/part-b-sectoral-guidance-chapters/1-energy/1-a-combustion/1-a-3-b-vi/view).
1695

1696 EMEP/EEA, 2016. Air Pollutant Emission Inventory Guidebook 2016. Road Transport Tyre and
1697 Brake Wear and Road Abrasion. EEA Report No 21/2016.
1698 [https://www.eea.europa.eu/publications/emep-eea-guidebook-2016/part-b-sectoral-guidance-
1699 chapters/1-energy/1-a-combustion/1-a-3-b-vi/view](https://www.eea.europa.eu/publications/emep-eea-guidebook-2016/part-b-sectoral-guidance-chapters/1-energy/1-a-combustion/1-a-3-b-vi/view)
1700

1701 EPA (US), 2020. Automotive Trends Report, EPA-420-R-21-003, Environmental Protection
1702 Agency, <https://www.epa.gov/automotive-trends>.
1703

1704 Fabretti, J.-F., Sauret, N., Gal, J.-F., Maria, P.-C., Schärar, U., 2009. Elemental characterization
1705 and source identification of PM_{2.5} using Positive Matrix Factorization: The Malraux road tunnel,
1706 Nice, France, *Atmos. Res.*, 94, 320-329.
1707
1708 Farwick zum Hagen, F.H., Mathissen, M., Grabiec, T., Hennicke, T., Rettig, M., Grochowicz, J.,
1709 Vogt, R., Benter, T., 2019a. On-road vehicle measurements of brake wear particle emissions,
1710 *Atmos. Environ.*, 217, 116943.
1711
1712 Farwick zum Hagen, F.H., Mathissen, M., Grabiec, T., Hennicke, T., Rettig, M., Grochowicz, J.,
1713 Vogt, R., Benter, T., 2019b. Study of brake wear particle emissions: impact of braking and cruising
1714 conditions, *Environ. Sci. Technol.*, 53, 5143-5150.
1715
1716 FAT, 2019. Air Quality Modelling on the Contribution of Brake Wear Emissions to Particulate
1717 Matter Concentrations Using a High-Resolution Brake Use Inventory, FAT-Schriftenreihe 318,
1718 Ricardo report to Verband der Automobilindustrie and Forschungsvereinigung Automobiltechnik,
1719 <https://www.vda.de/de/services/Publikationen/fat-schriftenreihe-318.html>.
1720
1721 Fieldhouse, J. D., Gelb, J., 2016. New developments of an on-vehicle brake pad waste collection
1722 system. *SAE Int. J. Passeng. Cars Mech. Syst.*, 9, 1245-1254.
1723
1724 Foitzik, M.-J., Unrau, H.-J., Gauterin, F., Dornhofer, J., Koch, T., 2018. Investigation of ultra fine
1725 particulate matter emission of rubber tires, *Wear*, 394-395, 87-95.
1726
1727 Furger, M., Minguillón, M. C., Yadav, V., Slowik, J. G., Hüglin, C., Fröhlich, R., Petterson, K.,
1728 Baltensperger, U., Prévôt, A. S. H., 2017. Elemental composition of ambient aerosols measured
1729 with high temporal resolution using an online XRF spectrometer, *Atmos. Meas. Tech.*, 10, 2061-
1730 2076.
1731
1732 Garg, B.D., Cadle, S.H., Mulawa, P.A., Groblicki, P.J., Laroo, C., Parr, G.A., 2000. Brake wear
1733 particulate matter emissions, *Environ. Sci. Technol.*, 34, 4463-4469.
1734
1735 Gaylard, A.P., Lynch, D., Amodeo, J., Amunugama, R., 2010. The Simulation of Brake Dust
1736 Deposition, presented at the 8th MIRA International Vehicle Aerodynamics Conference, 13-14
1737 October 2010.
1738
1739 Gehrig, R., Zeyer, K., Bukowiecki, N., Lienemann, P., Poulidakos, L.D., Furger, M., Buchmann,
1740 B., 2010. Mobile load simulators - A tool to distinguish between the emissions due to abrasion and
1741 resuspension of PM₁₀ from road surfaces, *Atmos. Environ.*, 44, 4937-4943.
1742
1743 Gietl, J.K., Lawrence, R., Thorpe, A.J. and Harrison, R.M., 2010. Identification of brake wear
1744 particles and derivation of a quantitative tracer for brake dust at a major road, *Atmos. Environ.*, 44,
1745 141-146.
1746
1747 Gonet, T., Maher, B.A., 2019. Airborne, vehicle-derived Fe-bearing nanoparticles in the urban
1748 environment: A review, *Environ. Sci. Technol.*, 53, 9970-9991.
1749
1750 Gramstat, S., Hascoet, M., Rocca-Serra, C., Adamczak, L., 2020. Potentials and challenges of a
1751 brake particle emission collecting system, SAE Technical Paper 2020-01-1635,
1752 <https://doi.org/10.4271/2020-01-1635>.
1753
1754 Grigoratos, T., Martini, G., 2015. Brake wear particle emissions: a review, *Environ. Sci., Pollut.*
1755 *Res.*, 22, 2491-2504.

- 1756 Grigoratos, T., Martini, G., 2014. Non-Exhaust Traffic Related Emissions. Brake and Tyre Wear
1757 PM, European Commission, Joint Research Centre, Institute of Energy and Transport, Ispra, Italy,
1758 doi:10.2790/21481.
1759
- 1760 Gustafsson, M., 2018. Review of Road Wear Emissions. A Review of Road Emission
1761 Measurement Studies: Identification of Gaps and Future Needs, in Non-Exhaust Emissions: An
1762 Urban Air Quality Problem for Public Health; Impact and Mitigation Measures, Amato., F. (Ed),
1763 Ch. 8, pp 161-181, Academic Press, 161-181, [https://doi.org/10.1016/B978-0-12-811770-5.00008-](https://doi.org/10.1016/B978-0-12-811770-5.00008-X)
1764 [X](https://doi.org/10.1016/B978-0-12-811770-5.00008-X).
- 1765 Gustafsson, M., Blomqvist, G., Gudmundsson, A., Dahl, A., Jonsson, P., Swietlicki, E., 2009.
1766 Factors influencing PM₁₀ emissions from road pavement wear, Atmos. Environ., 43, 4699-4702.
1767
- 1768 Gustafsson, M., Blomqvist, G., Gudmundsson, A., Dahl, A., Swietlicki, E., Bohgard, M., Lindbom,
1769 J., Ljungman, A., 2008. Properties and toxicological effects of particles from the interaction
1770 between tyres, road pavement and winter traction material, Sci. Tot. Environ., 393, 226-240.
1771 Hagino, H., Oyama, M., Sasaki, S., 2016. Laboratory testing of airborne brake wear particle
1772 emissions using a dynamometer system under urban city driving cycles, Atmos. Environ., 131, 269-
1773 278.
1774
- 1775 Harrison, R.M., Vu, T.V., Jafar, H., Shi, Z., 2021. More mileage in reducing urban air pollution
1776 from road traffic, Environ. Intl., 149, 106329.
1777
- 1778 Harrison, R.M., Jones, A.M., Gietl, J., Yin, J., Green, D.C., 2012. Estimation of the contributions
1779 of brake wear, tire wear, and resuspension to nonexhaust traffic particles derived from atmospheric
1780 measurements, Environ. Sci. Technol., 46, 6523-6529.
1781
- 1782 Harrison, R.M., Beddows, D.C.S., Dall'Osto, M., 2011. PMF analysis of wide-range particle size
1783 spectra collected on a major highway, Environ. Sci. Technol., 45, 5522–5528.
1784
- 1785 Hartmann, N.B., Hüffer, T., Thompson, R.C., Hassellöv, M., Verschoor, A., Dugaard, A.E., Rist,
1786 S., Karlsson, T., Brennholt, N., Cole, M., Herrling, M.P., Hess, M.C., Ivleva, N.P., Lusher, A.L.,
1787 Wagner, M., 2019. Are we speaking the same language? Recommendations for a definition and
1788 categorization framework for plastic debris, Environ. Sci. Technol., 53, 1039-1047.
1789
- 1790 Hascoet, M., Adamczak, L., 2020. At source brake dust collection system, Results Eng., 5, 100083.
1791
- 1792 Haynes, H.M., Taylor, K.G., Rothwell, J., Byrne, P., 2020. Characterisation of road-dust sediment
1793 in urban systems: a review of a global challenge, J. Soils Sediments, 20, 4194-4217.
1794
- 1795 Hicks, W., Beevers, S., Tremper, A. H., Stewart, G., Priestman, M., Kelly, F. J., Lanoisellé, M.,
1796 Lowry, D., Green, D. C., 2021. Quantification of Non-Exhaust Particulate Matter Traffic
1797 Emissions and the Impact of COVID-19 Lockdown at London Marylebone Road, Atmosphere, 12,
1798 10.3390/atmos12020190.
1799
- 1800 Hjortenkrans, D.S.T., Bergbäck, B. G., Häggerud, A.V., 2007. Metal emissions from brake linings
1801 and tires: Case studies of Stockholm, Sweden 1995/1998 and 2005, Environ. Sci. Technol., 41,
1802 5224-5230.
1803
- 1804 Hood, C., MacKenzie, I., Stocker, J., Johnson, K., Carruthers, D., Vieno, M., Doherty, R., 2018. Air
1805 quality simulations for London using a coupled regional-to-local modelling system, Atmos. Chem.
1806 Phys., 18, 11221-11245.

- 1807 Hood, C., Carruthers, D., Seaton, M., Stocker, J., Johnson, K., 2014. Urban Canopy Flow Field and
1808 Advanced Street Canyon Modelling in ADMS-Urban, in 16th International Conference on
1809 Harmonisation within Atmospheric Dispersion Modelling for Regulatory Purposes, Varna,
1810 Bulgaria, September 2014.
- 1811
- 1812 Hopke, P.K., Harrison, R.M., De Leeuw, F., Querol, X., 2018. Current State of Particulate Air
1813 Quality, in: Non-Exhaust Emission: An Urban Air Quality Problem for Public Health: Impact and
1814 mitigation measures, Amato, F. (Ed.), Academic Press, London.
- 1815
- 1816 Hulskotte, J.H.J., Roskam, G.D., Denier van der Gon, H.A.C., 2014. Elemental composition of
1817 current automotive braking materials and derived air emission factors, *Atmos. Environ.*, 99, 436-
1818 445.
- 1819
- 1820 Hussein, T., Johansson, C., Karlsson, H., Hansson, H.-C., 2008. Factors affecting non-tailpipe
1821 aerosol particle emissions from paved roads: On-road measurements in Stockholm, Sweden,
1822 *Atmos. Environ.*, 42, 688-702.
- 1823
- 1824 Hwang, H.-M., Fiala, M.J., Wade, T.L., Park, D., 2019. Review of pollutants in urban road dust:
1825 Part II. Organic contaminants from vehicles and road management, *Int. J. Urban Sci.*, 23, 445-463.
- 1826
- 1827 Hwang, H.-M., Fiala, M.J., Park, D., Wade, T.L., 2016. Review of pollutants in urban road dust
and stormwater runoff: part 1. Heavy metals released from vehicles, *Int. J. Urban Sci.*, 20, 334-360.
- 1828
- 1829 Iijima, A., Sato, S., Fujitani, Y., Fujimori, E., Saito, Y., Tanabe, K., Ohara, T., Kozawa, K., Furuta,
1830 N., 2009. Clarification of the predominant emission sources of antimony in airborne particulate
1831 matter and estimation of their effects on the atmosphere in Japan, *Environ. Chem.*, 6, 122-132.
- 1832
- 1833 Iijima, A., Sato, K., Yano, K., Kato, M., Kozawa, K., Furuta, N., 2008. Emission factor for
1834 antimony in brake abrasion dusts as one of the major atmospheric antimony sources, *Environ. Sci.*
1835 *Technol.*, 42, 2937-2942.
- 1836
- 1837 Jeong, C.-H., Wang, J.M., Hilker, N., Debosz, J., Sofowote, U., Su, Y., Noble, M., Healy, R. M.,
1838 Munoz, T., Dabek-Zlotorzynska, E., Celoz, V., White, L., Audette, C., Herod, D., Evans, G. J., 2019.
1839 Temporal and spatial variability of traffic-related PM_{2.5} sources: Comparison of exhaust and non-
1840 exhaust emissions, *Atmos. Environ.*, 198, 55-69.
- 1841
- 1842 Johansson, C., Denby, B.R., Sundvor, I., Kauhaniemi, M., Härkönen, J., Kukkonen, J., Karppinen,
1843 A., Kangas, L., Omstedt, G., Ketzler, M., Massling, A., Pirjola, L., Norman, M., Gustafsson, M.,
1844 Blomqvist, G., Bennet, C., Kupiainen, K., Karvosenoja, N., 2012. NORTRIP, NON-exhaust Road
1845 TRaffic Induced Particle emissions: Development of a model for assessing the effect on air quality
1846 and exposure. ITM Report 212, Department of Applied Environmental Science (ITM), Stockholm
1847 University, <http://vti.diva-portal.org/smash/get/diva2:674122/FULLTEXT01.pdf>.
- 1848
- 1849 Joo, B.S., Jara, D.C., Seo, H.J., Jang, H., 2020. Influences of the average molecular weight of
1850 phenolic resin and potassium titanate morphology on particulate emissions from brake linings,
1851 *Wear*, 450-451, 203243.
- 1852
- 1853 Joo, B.S., Chang, Y.H., Seo, H.J., Jang, H., 2019. Effects of binder resin on tribological properties
1854 and particle emission of brake linings, *Wear*, 434-435, 202995.
- 1855
- 1856 Karanasiou, A., Moreno, T., Amato, F., Lumberras, J., Narros, A., Borge, R., Tobias, A., Boldo, E.,
Linares, C., Pey, J., Reche, C., Alastuey, A., Querol, X., 2011. Road dust contribution to PM levels

1857 – Evaluation of the effectiveness of street washing activities by means of Positive Matrix
1858 Factorisation, *Atmos. Environ.*, 45, 2193-2201.

1859

1860 Kennedy, P., Gadd, J., Moncrieff, I., 2002. Emission Factors for Contaminants Released by Motor
1861 Vehicles in New Zealand, Ministry of Transport, New Zealand.

1862

1863 Khan, R.K., Strand, M.A., 2018. Road dust and its effect on human health: a literature review,
1864 *Epidemiol. Health*, 40, e2018013-2018010, 10.4178/epih.e2018013.

1865

1866 Kim, S.-H., Shim, W., Kwon, S.-U., Lee, J.-J., Seo, M.-W., Kim, J.-K., Pee, J.-H., Kim, J.-Y., 2020.
1867 The impact of composition in non-steel and low-steel type friction materials on airborne brake wear
1868 particulate emission, *Tribol. Lett.*, 68, 118, <https://doi.org/10.1007/s11249-020-01361-2>.

1869

1870 Kim, G., Lee, S., 2018. Characteristics of tire wear particles generated by a tire simulator under
1871 various driving conditions, *Environ. Sci. Technol.*, 52, 12153-12161.

1872

1873 Klockner, P., Seiwert, B., Weyrauch, S., Escher, B.I., Reemtsma, T., Wagner, S., 2021.
1874 Comprehensive characterization of tire and road wear particles in highway tunnel road dust by use
1875 of size and density fractionation, *Chemosphere*, 279, 130530.

1876

1877 Knight, L.J., Parker-Jurd, F.N.F., Al-Sid-Cheikh, M., Thompson, R.C., 2020. Tyre wear particles:
1878 an abundant yet widely unreported microplastic?, *Environ. Sci. Pollut. Res.*, 27, 18345-18354.

1879

1880 Kole, P.J., Löhr, A.J., Van Belleghem, F.G.A.J., Ragas, A.M.J., 2017. Wear and tear of tyres: A
1881 stealthy source of microplastics in the environment, *Int. J. Environ. Res. Public Health*, 14, 1265,
1882 <https://www.mdpi.com/1660-4601/14/10/1265>

1883

1884 Kreider, M.L., Panko, J.M., McAtee. B.L., Sweet. L.I., Finley. B.L., 2010. Physical and chemical
1885 characterization of tire-related particles: Comparison of particles generated using different
1886 methodologies, *Sci. Tot. Environ.*, 408, 652-659.

1887

1888 Kukutschová, J., Filip, P., 2018. Review of Brake Wear Emissions. A Review of Brake Emission
1889 Measurement Studies: Identification of Gaps and Future Needs, in *Non-Exhaust Emissions. An
1890 Urban Air Quality Problem for Public Health; Impact and Mitigation Measures*, Amato., F. (Ed),
1891 Ch. 6, pp 123-146, <https://doi.org/10.1016/B978-0-12-811770-5.00006-6>.

1892

1893 Kukutschová, J., Moravec, P., Tomášek, V., Matějka, V., Smolík, J., Schwarz, J., Seidlerová, J.,
1894 Šafářová, K., Filip, P.. 2011. On airborne nano/micro-sized wear particles released from low-
1895 metallic automotive brakes, *Environ. Pollut.*, 159, 998-1006.

1896

1897 Kumata, H., Mori, M., Takahashi, S., Takamiya, S., Tsuzuki, M., Uchida, T., and Fujiwara, K.,
1898 2011. Evaluation of Hydrogenated Resin Acids as Molecular Markers for Tire-wear Debris in
1899 Urban Environments, *Environ. Sci. Technol.*, 45, 9990-9997.

1900

1901 Kupiainen, K., 2007. Road dust from pavement wear and traction sanding, *Monographs Boreal
1902 Environ. Res.*, 26, Finnish Environmental Research Institute, Finland, Helsinki,
1903 https://helda.helsinki.fi/bitstream/handle/10138/39334/BERMon_26.pdf.

1904

1905 Kwak, J., Lee, S., Lee, S., 2014. On-road and laboratory investigations on non-exhaust ultrafine
1906 particles from the interaction between the tire and road pavement under braking conditions, *Atmos.
1907 Environ.*, 97, 195-205.

1908

1909 Kwak, J.-H., Kim, H., Lee, J., Lee, S., 2013. Characterization of non-exhaust coarse and fine
1910 particles from on-road driving and laboratory measurements, *Sci. Total Environ.*, 458-460, 273-282,
1911 doi:<https://doi.org/10.1016/j.scitotenv.2013.04.040>.
1912

1913 Ladonin, D.V., 2018. Platinum-group elements in soils and street dust of the southeastern
1914 administrative district of Moscow, *Eurasian Soil Sci.*, 51, 268-276.
1915

1916 Lanzerstorfer, C., 2020. Toward more intercomparable road dust studies, *Crit. Rev. Environ. Sci.*
1917 *Technol.*, 10.1080/10643389.2020.1737472.
1918 Lanzerstorfer, C., 2018. Heavy metals in the finest size fractions of road-deposited sediments,
1919 *Environ. Pollut.*, 239, 522-531.
1920

1921 Lawrence, S., Sokhi, R., Ravindra, K., Mao, H., Prain, H. D., Bull, I.D., 2013. Source
1922 apportionment of traffic emissions of particulate matter using tunnel measurements, *Atmos.*
1923 *Environ.*, 77, 548-557.
1924

1925 Lee, S., Kwak, J., Kim, H., Lee, J., 2013. Properties of roadway particles from interaction between
1926 the tire and road pavement, *Int. J. Automot. Technol.*, 14, 163-173.
1927

1928 Lenschow, P., Abraham, H.J., Kutzner, K., Lutz, M., Preuß, J.D., Reichenbacher, W., 2001. Some
1929 ideas about the sources of PM₁₀, *Atmos. Environ.*, 35, S23-S33.
1930

1931 Liao, C., Kim, U.-J., Kannan, K., 2018. A Review of Environmental Occurrence, Fate, Exposure,
1932 and Toxicity of Benzothiazoles, *Environ. Sci. Technol.*, 52, 5007-5026.
1933

1934 Lin, L., Lee, M. L., Eatough, D.J., 2010. Review of recent advances in detection of organic
1935 markers in fine particulate matter and their use for source apportionment, *JAWMA*, 60, 3-25, 2010.
1936

1937 Lo, C.-Y., Khodadadi, H., Hua, M., De Rosa, I.M., Carlson, L., Kavazanjian Jr, E., He, X., 2020.
1938 Durable and ductile double-network material for dust control, *Geoderma*, 361, 114090.
1939

1940 Lough, G.C., Schauer, J.J., Park, J.-S., Shafer, M.M., Deminter, J.T., Weinstein, J.P., 2005.
1941 Emissions of metals associated with motor vehicle roadways, *Environ. Sci. Technol.*, 39, 826-836.
1942

1943 Lyu, Y., Leonardi, M., Wahlstrom, J., Gialanella, S., Olofsson, U., 2020. Friction, wear and
1944 airborne particle emission from Cu-free brake materials, *Tribol. Int.*, 141, 105959.
1945

1946 Maeaba, W., Kumari, R., Prasad, S., 2020. Spectroscopic assessment of heavy metals pollution in
1947 roadside soil and road dust: a review, *Appl. Spectrosc. Rev.*, 10.1080/05704928.2020.1835940.
1948

1949 Mathissen, M., Grochowicz, J., Schmidt, C., Vogt, R., Farwick zum Hagen, F.H., Grabiec, T.,
1950 Steven, H., Grigoratos, T., 2018. A novel real-world braking cycle for studying brake wear particle
1951 emissions, *Wear*, 414-415, 219-226.
1952

1953 Mathissen, M., Scheer, V., Vogt, R., Benter, T., 2011. Investigation on the potential generation of
1954 ultrafine particles from the tire-road interface, *Atmos. Environ.*, 45, 6172-6179.
1955

1956 Memorandum of understanding on copper mitigation in watersheds and waterways between...,
1957 2015
1958 https://www.epa.gov/sites/production/files/2015-11/documents/copper_brakepads_mou.pdf
1959

- 1960 Mengistu, D., Nilsen, V., Heistad, A., Kvaal, K., 2019. Detection and quantification of tire particles
1961 in sediments using a combination of simultaneous thermal analysis, fourier transform infra-red, and
1962 parallel factor analysis, *Int. J. Env. Res. Pub. Health*, 16, 3444, 2019.
- 1963
- 1964 Metre, P.C.V., Mahler, B.J., Wilson, J.T., 2009. PAHs Underfoot: Contaminated Dust from Coal-
1965 Tar Sealcoated Pavement is Widespread in the United States, *Environ. Sci. Technol.*, 43, 20-25.
- 1966
- 1967 Monks, P., Allan, J., Carruthers, D., Carslaw, D., Fuller, G., Harrison, R., Heal, M., Lewis, A.,
1968 Nemitz, E., Reeves, C., Williams, M., Fowler, D., Marnner, B., Williams, A., Moller, S., Maggs, R.,
1969 Murrells, T., Quincey, P., Willis, P., 2019. Non-Exhaust Emissions from Road Traffic, Report
1970 prepared for Department for Environment, Food and Rural Affairs; Scottish Government, Welsh
1971 Government and Department of Environment in Northern Ireland.
- 1972
- 1973 Murphy, D.M., 2007. The design of single particle laser mass spectrometers, *Mass Spectrom. Rev.*,
1974 26, 150-165.
- 1975
- 1976 NAEI, 2020. UK National Atmospheric Emissions Inventory, <http://naei.beis.gov.uk/>.
- 1977
- 1978 Nielsen, M., Winther, M., Boll, J., Mette, I., Mikkelsen, H., 2003. Danish emission inventory for
1979 particulate matter (PM), Research Notes from NERI No. 189,
1980 https://www.dmu.dk/1_viden/2_Publikationer/3_arbrapporter/rapporter/AR189.pdf.
- 1981
- 1982 NIOSH, 2000. Health effects of occupational exposure to asphalt, National Institute for
1983 Occupational Safety and Health, DHHS (NIOSH) Publication No. 2001-110
1984 <https://www.cdc.gov/niosh/docs/2001-110/default.html>.
- 1985 Nolte, C.G., Schauer, J.J., Cass, G.R., Simoneit, B.R.T., 2002. Trimethylsilyl Derivatives of
1986 Organic Compounds in Source Samples and in Atmospheric Fine Particulate Matter, *Environ. Sci.*
1987 *Technol.*, 36, 4273-4281.
- 1988
- 1989 Nosko, O. and Olofsson, U., 2017. Effective density of airborne wear particles from car brake
1990 materials, *J. Aerosol Sci.*, 107, 94-106,
- 1991
- 1992 Nosko, O., Vanhanen, J. Olofsson, U., 2017. Emission of 1.3-10 nm airborne particles from brake
1993 materials, *Aerosol Sci. Technol.*, 51, 91-96.
- 1994
- 1995 Noziere, B., Kaberer, M., Claeys, M., Allan, J., D'Anna, B., Decesari, S., Finessi, E., Glasius, M.,
1996 Grgic, I., Hamilton, J. F., Hoffmann, T., Iinuma, Y., Jaoui, M., Kahno, A., Kampf, C. J., Kourtchev,
1997 I., Maenhaut, W., Marsden, N., Saarikoski, S., Schnelle-Kreis, J., Surratt, J. D., Szidat, S.,
1998 Szmigielski, R., and Wisthaler, A., 2015. The molecular identification of organic compounds in the
1999 atmosphere: State of the art and challenges, *Chem. Rev.*, 115, 3919-3983.
- 2000
- 2001 OECD, 2020. Non-exhaust Particulate Emissions from Road Transport: An Ignored Environmental
2002 Policy Challenge, OECD Publishing, Paris, <https://doi.org/10.1787/4a4dc6ca-en>.
- 2003
- 2004 Ondráček, J., Schwarz, J., Ždímal, V., Andělová, L., Vodička, P., Bízek, V., Tsai, C. J., Chen, S. C.,
2005 Smolík, J., 2011. Contribution of the road traffic to air pollution in the Prague city (busy speedway
2006 and suburban crossroads), *Atmos. Environ.*, 45, 5090-5100.
- 2007
- 2008 Österle, W., Griepentrog, M., Gross, T., Urban, I., 2001. Chemical and microstructural changes
2009 induced by friction and wear of brakes, *Wear*, 251, 1469-1476.
- 2010

2011 Ostermeyer, G.P., Muller, M., 2008. New insights into the tribology of brake systems, Proc. Inst.
2012 Mech. Engrs. 222(D): J. Auto. Eng., 222, 1167-1200.
2013
2014 Paatero, P., Tapper, U., 1994. Positive matrix factorization: A non-negative factor model with
2015 optimal utilization of error estimates of data values, Environmetrics, 5, 111-126.
2016
2017 Paatero, P., 1999. The multilinear engine - A table-driven, least squares program for solving
2018 multilinear problems, including the n-way parallel factor analysis model, J. Comput. Graph. Stat., 8,
2019 854-888.
2020
2021 Paatero, P., 1997. Least squares formulation of robust non-negative factor analysis, Chemometr.
2022 Intell. Lab., 37, 23-35.
2023
2024 Padoan, E., Romè, C., Ajmone-Marsan, F., 2017. Bioaccessibility and size distribution of metals in
2025 road dust and roadside soils along a peri-urban transect, Sci. Tot. Environ., 601-602, 89-98.
2026
2027 Panko, J.M., Hitchcock, K.M., Fuller, G.W., Green, D., 2019. Evaluation of tire wear contribution
2028 to PM_{2.5} in urban environments, Atmosphere, 10, 99, 10.3390/atmos10020099.
2029
2030 Panko, J.M., Chu, J., Kreider, M.L., Unice, K.M., 2013. Measurement of airborne concentrations of
2031 tire and road wear particles in urban and rural areas of France, Japan, and the United States, Atmos.
2032 Environ., 72, 192-199.
2033
2034 Pant, P., Shukla, A., Kohl, S.D., Chow, J.C., Watson, J.G., Harrison, R.M., 2015. Characterization
2035 of ambient PM_{2.5} at a pollution hotspot in New Delhi, India and inference of sources, Atmos.
2036 Environ., 109, 178-189.
2037
2038 Pant, P., Harrison, R.M., 2013. Estimation of the contribution of road traffic emissions to
2039 particulate matter concentrations from field measurements: A review, Atmos. Environ., 77, 78-97.
2040
2041 Park, I., Kim, H., Lee, S., 2018. Characteristics of tire wear particles generated in a laboratory
2042 simulation of tire/road contact conditions, J. Aerosol Sci., 124, 30-40.
2043
2044 Park, I., Lee, J., Lee, S., 2017. Laboratory study of the generation of nanoparticles from tire tread,
2045 Aerosol Sci. Technol., 51, 188-197.
2046
2047 Passant, N.R., 2002. Speciation of UK emissions of non-methane volatile organic compounds.
2048 NAEI Report AEAT/ENV/R/0545 prepared for DETR Air and Environmental Quality Division,
2049 February 2002, [https://uk-](https://uk-air.defra.gov.uk/assets/documents/reports/empire/AEAT_ENV_0545_final_v2.pdf)
2050 [air.defra.gov.uk/assets/documents/reports/empire/AEAT_ENV_0545_final_v2.pdf](https://uk-air.defra.gov.uk/assets/documents/reports/empire/AEAT_ENV_0545_final_v2.pdf)
2051
2052 Pay, M.T., Jiménez-Guerrero, P., Baldasano J.M., 2011. Implementation of resuspension from
2053 paved roads for the improvement of CALIOPE air quality system in Spain. Atmos. Environ., 45,
2054 802-807.
2055
2056 Pérez, N., Pey, J., Cusack, M., Reche, C., Querol, X., Alastuey, A., Viana, M., 2010. Variability of
2057 particle number, black carbon, and PM₁₀, PM_{2.5}, and PM₁ levels and speciation: Influence of road
2058 traffic emissions on urban air quality, Aerosol Sci. Technol., 44, 487-499.
2059
2060 Perricone, G., Matejka, V., Alemani, M., Valota, G., Bonfanti, A., Ciotti, A., Olofsson, U.,
2061 Söderberg, A., Wahlström, J., Nosko, O., Straffelini, G., Gialanella, S., Ibrahim, M., 2018. A
2062 concept for reducing PM₁₀ emissions for car brakes by 50%, Wear, 396-397, 135-145.

2063 Pipalatkhar, P., Khaparde, V.V., Gajghate, D.G., Bawase, M.A., 2014. Source Apportionment of
2064 PM_{2.5} Using a CMB Model for a Centrally Located Indian City, *Aerosol Air Qual. Res.*, 14, 1089-
2065 1099.

2066

2067 Pischinger, R., Rauterberg-Wulff, A., 1999. Determination of the emission factors for tyre wear
2068 particles up to 10 microns by tunnel measurements. In *Proceedings of the 8th International*
2069 *Symposium Transport and Air Pollution including COST 319 Final Conference, Graz, Austria, 31*
2070 *May-2 June.*

2071

2072 Plachá, D., Vaculík, M., Mikeska, M., Dutko, O., Peikertová, P., Kukutschová, J., Mamulová
2073 Kutlákova, K., Růžicková, J., Tomášek, V., Filip, P., 2017. Release of volatile organic compounds
2074 by oxidative wear of automotive friction materials, *Wear*, 376-377, 705-716.

2075

2076 Pratt, K.A., Prather, K. A., 2012. Mass spectrometry of atmospheric aerosols-Recent developments
2077 and applications. Part II: On-line mass spectrometry techniques, *Mass. Spectrom. Rev.*, 31, 17-48.

2078

2079 Querol, X., Alastuey, A., Moreno, T., Viana, M. M., Castillo, S., Pey, J., Rodríguez, S., Artiñano,
2080 B., Salvador, P., Sánchez, M., Garcia Dos Santos, S., Herce Garraleta, M. D., Fernandez-Patier, R.,
2081 Moreno-Grau, S., Negral, L., Minguillón, M. C., Monfort, E., Sanz, M. J., Palomo-Marín, R.,
2082 Pinilla-Gil, E., Cuevas, E., de la Rosa, J., Sánchez de la Campa, A., 2008. Spatial and temporal
2083 variations in airborne particulate matter (PM₁₀ and PM_{2.5}) across Spain 1999–2005, *Atmos.*
2084 *Environ.*, 42, 3964-3979.

2085

2086 Rai, P., Furger, M., Slowik, J.G., Canonaco, F., Fröhlich, R., Hüglin, C., Minguillón, M. C.,
2087 Petterson, K., Baltensperger, U., Prévôt, A.S.H., 2020. Source apportionment of highly time-
2088 resolved elements during a firework episode from a rural freeway site in Switzerland, *Atmos.*
2089 *Chem. Phys.*, 20, 1657-1674.

2090

2091 Rai, P., Slowik, J. G., Furger, M., El Haddad, I., Visser, S., Tong, Y., Singh, A., Wehrle, G.,
2092 Kumar, V., Tobler, A. K., Bhattu, D., Wang, L., Ganguly, D., Rastogi, N., Huang, R. J., Necki, J.,
2093 Cao, J., Tripathi, S. N., Baltensperger, U., Prévôt, A.S.H., 2021. Highly time-resolved
2094 measurements of element concentrations in PM₁₀ and PM_{2.5}: Comparison of Delhi, Beijing,
2095 London, and Krakow, *Atmos. Chem. Phys.*, 21, 717-730.

2096

2097 Raisanen, M., Kupiainen, A.K., Tervahattu, A.H., 2003. The effect of mineralogy, texture and
2098 mechanical properties of anti-skid and asphalt aggregates on urban dust, *Bull. Eng. Geol. Environ.*,
2099 62, 359-368.

2100

2101 Rattanasom, N., Saowapark, T., and Deeprasertkul, C., 2007. Reinforcement of natural rubber with
2102 silica/carbon black hybrid filler, *Polym. Test.*, 26, 369-377.

2103

2104 Rebotier, T.P., Prather, K.A., 2007. Aerosol time-of-flight mass spectrometry data analysis: a
2105 benchmark of clustering algorithms, *Anal. Chim. Acta.*, 585, 38-54.

2106

2107 Rogge, W.F., Hildemann, L.M., Mazurek, M.A., Cass, G.R., Simoneit, B.R.T., 1993. Sources of
2108 fine organic aerosol. 3. Road dust, tire debris, and organometallic brake lining dust: roads as
2109 sources and sinks, *Environ. Sci. Technol.*, 27, 1892-1904.

2110

2111 Rosselot, K.S., 2006. Copper Released from Brake Lining Wear in the San Fransisco Bay Area,
2112 Report prepared for the Brake Pad Partnership, January 2006,
2113 <http://www.suscon.org/pdfs/bpp/pdfs/BrakeSourcesReportFinal01-30-06a.pdf>.

2114

2115 Roubicek, V., Raclavska, H., Juchelkova, D., Filip, P., 2008. Wear and environmental aspects of
2116 composite materials for automotive braking industry, *Wear*, 265, 167-175.
2117
2118 Sadiktsis, I., Bergvall, C., Johansson, C., Westerholm, R., 2012. Automobile tires - A Potential
2119 source of highly carcinogenic dibenzopyrenes to the environment, *Environ. Sci. Technol.*, 46, 3326-
2120 3334.
2121
2122 Saikawa, E., Kim, H., Zhong, M., Avramov, A., Zhao, Y., Janssens-Maenhout, G., Kurokawa, J.,
2123 Klimont, Z., Wagner, F., Naik, V., Horowitz, L.W., Zhang, Q., 2017. Comparison of emissions
2124 inventories of anthropogenic air pollutants and greenhouse gases in China, *Atmos. Chem. Phys.*, 17,
2125 6393-6421.
2126
2127 Sanders, P.G., Xu, N., Dalka, T.M., Maricq, M., 2003. Airborne brake wear debris: Size
2128 distributions, composition, and a comparison of dynamometer and vehicle tests, *Environ. Sci.*
2129 *Technol.*, 37, 4060-4069.
2130
2131 Schrewe, K., Steigert, S., Neumann, P., 2012. Partial flow sintered metal filter as part of a tier 4
2132 emission reduction strategy, *SAE Int. J. Fuels and Lubr.*, 5, 1291-1299.
2133 Seo, H., Joo, B., Park, J., Kim, Y.C., Lee, J.J., Jang, H., 2021. Effect of disc material on particulate
2134 matter emissions during high temperature braking, *Tribol. Int.*, 154, 106713.
2135
2136 Simpson, D., Fargerli, H., Jonson, J. E., Tsyro, S. and Wind, P., 2003. Transboundary acidification,
2137 eutrophication and ground level ozone in Europe. Part 1. Unified EMEP model description,
2138 EMEP/MSC-W Report 1/2003, The Norwegian Meteorological Institute, Oslo, Norway.
2139
2140 Sjodin, A., Ferm, M., Björk, A., Rahmberg, M., Gudmundsson, A., Swietlicki, E., Johansson, C.,
2141 Gustafsson, M., Blomqvist, G., 2010. Wear particles from road traffic - a field, laboratory and
2142 modelling study - Final Report, IVL Swedish Environmental Research Institute Ltd., Göteborg,
2143 Sweden.
2144
2145 Smit, R., 2014. Australian Motor Vehicle Emission Inventory for the National Pollutant Inventory
2146 (NPI).
2147
2148 Smolders, E., Degryse, F., 2002. Fate and effect of zinc from tire debris in soil, *Environ. Sci.*
2149 *Technol.*, 36, 3706-3710.
2150
2151 Sommer, F., Dietze, V., Baum, A., Sauer, J., Gilge, S., Maschowski, C., Gieré, R., 2018. Tire
2152 abrasion as a major source of microplastics in the environment, *Aerosol Air Qual. Res.*, 18, 2014-
2153 2028.
2154
2155 Song, F., Gao, Y., 2011. Size distributions of trace elements associated with ambient particular
2156 matter in the affinity of a major highway in the New Jersey–New York metropolitan area, *Atmos.*
2157 *Environ.*, 45, 6714-6723.
2158
2159 Suleiman, A., Tight, M.R., Quinn, A.D., 2019. Applying machine learning methods in managing
2160 urban concentrations of traffic-related particulate matter (PM₁₀ and PM_{2.5}), *Atmos. Pollut. Res.*, 10,
2161 134-144.
2162
2163 Tanner, P.A., Ma, H.-L., Yu, P.K.N., 2008. Fingerprinting metals in urban street dust of Beijing,
2164 Shanghai, and Hong Kong, *Environ. Sci. Technol.*, 42, 7111-7117.
2165

2166 Tarasiuk, W., Golak, K., Tsybrii, Y., Nosko, O., 2020. Correlations between the wear of car brake
2167 friction materials and airborne wear particle emissions, *Wear*, 456-457, 203361.
2168

2169 ten Broeke, H., Hulskotte, J., Denier van der Gon, H., 2008. Netherlands Emission Inventory:
2170 Road traffic tyre wear, TNO Built Environment and Geosciences,
2171 [http://www.emissieregistratie.nl/ERPUBLIEK/documenten/Water/Factsheets/English/Road%20surf](http://www.emissieregistratie.nl/ERPUBLIEK/documenten/Water/Factsheets/English/Road%20surface%20wear.pdf)
2172 [ace%20wear.pdf](http://www.emissieregistratie.nl/ERPUBLIEK/documenten/Water/Factsheets/English/Road%20surface%20wear.pdf).
2173

2174 Thorpe, A., Harrison, R.M., 2008. Sources and properties of non-exhaust particulate matter from
2175 road traffic: A review, *Sci. Tot. Environ.*, 400, 270-282.

2176 Timmers, V., Achten, P., 2016. Non-exhaust PM emissions from electric vehicles, *Atmos.*
2177 *Environ.*, 134, 10-17.
2178

2179 Tremper, A.H., Font, A., Priestman, M., Hamad, S.H., Chung, T.C., Pribadi, A., Brown, R.J.C.,
2180 Goddard, S.L., Grassineau, N., Petterson, K., Kelly, F.J., Green, D.C., 2018. Field and laboratory
2181 evaluation of a high time resolution x-ray fluorescence instrument for determining the elemental
2182 composition of ambient aerosols, *Atmos. Meas. Tech.*, 11, 3541-3557.
2183

2184 Unice, K. M., Kreider, M. L., and Panko, J. M., 2012. Use of a deuterated internal standard with
2185 pyrolysis-GC/MS dimeric marker analysis to quantify tire tread particles in the environment, *Int. J.*
2186 *Env. Res. Pub. Health*, 9, 4033-4055.
2187

2188 USEPA, 2020a. Brake and Tire Wear Emissions from Onroad Vehicles in MOVES3, Report of the
2189 United States Environmental Protection Agency, EPA-420-R-20-014, November 2020,
2190 <https://nepis.epa.gov/Exe/ZyPDF.cgi/P1010M43.PDF?Dockey=P1010M43.PDF>.
2191

2192 USEPA, 2020b. United States Environmental Protection Agency, CMAQ (Version 5.3.2)
2193 [Software], <https://doi.org/10.5281/zenodo.4081737>.
2194

2195 USEPA, 2017. United States Environmental Protection Agency, 2017 National Emissions Inventory
2196 (NEI) Data, available at [https://www.epa.gov/air-emissions-inventories/2017-national-emissions-](https://www.epa.gov/air-emissions-inventories/2017-national-emissions-inventory-nei-data)
2197 [inventory-nei-data](https://www.epa.gov/air-emissions-inventories/2017-national-emissions-inventory-nei-data)
2198

2199 USEPA, 2011. Emission Factor Documentation for AP-42, Section 13.2.1: Paved Roads.
2200 Measurement Policy Group, Office of Air Quality Planning and Standards. U.S. Environmental
2201 Protection Agency, January 2011. <https://www3.epa.gov/ttn/chief/ap42/ch13/bgdocs/b13s0201.pdf>.
2202

2203 Varrica, D., Bardelli, F., Dongarrà, G., and Tamburo, E., 2013. Speciation of Sb in airborne
2204 particulate matter, vehicle brake linings, and brake pad wear residues, *Atmos. Environ.*, 64, 18-24.

2205 Vieno, M., Heal, M. R., Williams, M. L., Carnell, E. J., Nemitz, E., Stedman, J. R., Reis, S., 2016.
2206 The sensitivities of emissions reductions for the mitigation of UK PM_{2.5}, *Atmos. Chem. Phys.*, 16,
2207 265-276.
2208

2209 Visser, S., Slowik, J. G., Furger, M., Zotter, P., Bukowiecki, N., Canonaco, F., Flechsig, U., Appel,
2210 K., Green, D. C., Tremper, A.H., Young, D.E., Williams, P.I., Allan, J.D., Coe, H., Williams, L.R.,
2211 Mohr, C., Xu, L., Ng, N L., Nemitz, E., Barlow, J.F., Halios, C.H., Fleming, Z.L., Baltensperger,
2212 U., Prévôt, A.S.H., 2015. Advanced source apportionment of size-resolved trace elements at
2213 multiple sites in London during winter, *Atmos. Chem. Phys.*, 15, 11291-11309.
2214

2215 Vu, T.V., Delgado-Saborit, J.M., Harrison, R. M., 2015. Review: Particle number size distributions
2216 from seven major sources and implications for source apportionment studies, *Atmos. Environ.*, 122,
2217 114-132.

2218 Wahlin, P., Berkowicz, R., Palmgren, F., 2006. Characterisation of traffic-generated particulate
2219 matter in Copenhagen, *Atmos. Environ.*, 40, 2151-2159.
2220
2221 Wahlstrom, J., Olander, L., Olofsson, U., 2010a. Size, shape, and elemental composition of
2222 airborne wear particles from disc brake materials, *Tribol. Lett.*, 38, 15-24.
2223
2224 Wahlstrom, J., Söderberg, A., Olander, L., Jansson, A., Olofsson, U., 2010b. A pin-on-disc
2225 simulation of airborne wear particles from disc brakes, *Wear*, 268, 763-769.
2226
2227 Wahlstrom, J., Söderberg, A., Olander, L., Olofsson, U., 2009. A disc brake test stand for
2228 measurement of airborne wear particles, *Lubr. Sci.*, 21, 241-252.
2229
2230 Wang, Y., Li, X., 2012. Health risk of platinum group elements from automobile catalysts,
2231 *Procedia Eng.*, 45, 1004-1009.
2232
2233 Wei, L., Choy, Y.S., Cheung, C.S., Jin, D., 2020. Tribology performance, airborne particle
2234 emissions and brake squeal noise of copper-free friction materials, *Wear*, 448-449, 203215.
2235
2236 Weinbruch, S., Worrigen, A., Ebert, M., Scheuven, D., Kandler, K., Pfeffer, U., Bruckmann, P.,
2237 2014. A quantitative estimation of the exhaust, abrasion and resuspension components of
2238 particulate traffic emissions using electron microscopy, *Atmos. Environ.*, 99, 175-182.
2239
2240 Wiedensohler, A., Birmili, W., Nowak, A., Sonntag, A., Weinhold, K., Merkel, M., Wehner, B.,
2241 Tuch, T., Pfeifer, S., Fiebig, M., Fjaraa, A. M., Asmi, E., Sellegri, K., Depuy, R., Venzac, H.,
2242 Villani, P., Laj, P., Aalto, P., Ogren, J. A., Swietlicki, E., Williams, P., Roldin, P., Quincey, P.,
2243 Huglin, C., Fierz-Schmidhauser, R., Gysel, M., Weingartner, E., Riccobono, F., Santos, S.,
2244 Gruning, C., Faloon, K., Beddows, D., Harrison, R.M., Monahan, C., Jennings, S.G., O'Dowd,
2245 C.D., Marinoni, A., Horn, H.G., Keck, L., Jiang, J., Scheckman, J., McMurry, P.H., Deng, Z., Zhao,
2246 C.S., Moerman, M., Henzing, B., de Leeuw, G., Loschau, G., Bastian, S., 2012. Mobility particle
2247 size spectrometers: harmonization of technical standards and data structure to facilitate high quality
2248 long-term observations of atmospheric particle number size distributions, *Atmos. Meas. Tech.*, 5,
2249 657-685.
2250
2251 Winkler, S.L., Anderson, J.E., Garza, L., Ruona, W.C., Vogt, R., Wallington, T.J., 2018. Vehicle
2252 criteria pollutant (PM, NO_x, CO, HCs) emissions: how low should we go?, *npj Clim. Atmos. Sci.*,
2253 26, <https://doi.org/10.1038/s41612-018-0037-5>.
2254
2255 Zereini, F., Alsenz, H., Wiseman, C.L.S., Püttmann, W., Reimer, E., Schleyer, R., Bieber, E.,
2256 Wallasch, M., 2012. Platinum group elements (Pt, Pd, Rh) in airborne particulate matter in rural vs.
2257 urban areas of Germany: Concentrations and spatial patterns of distribution, *Sci. Tot. Environ.*, 416,
2258 261-268.
2259
2260 Zhang, J., Peng, J., Song, C., Ma, C., Men, Z., Wu, J., Wu, L., Wang, T., Zhang, X., Tao, S., Gao,
2261 S., Hopke, P.K., Mao, H., 2020. Vehicular non-exhaust particulate emissions in Chinese
2262 megacities: Source profiles, real-world emission factors, and inventories, *Environ. Pollut.*, 266,
2263 115268.
2264
2265 Zhang, J., Zhang, X., Wu, L., Wang, T., Zhao, J., Zhang, Y., Men, Z., Mao, H., 2018. Occurrence
2266 of benzothiazole and its derivatives in tire wear, road dust, and roadside soil, *Chemosphere*, 201, 310-
2267 317.
2268

2269 Zhu, D., Kuhns, H.D., Gillies, J.A., Etyemezian, V., Brown, S., Gertler, A.W., 2012. Analysis of
2270 the effectiveness of control measures to mitigate road dust emissions in a regional network, *Transp.*
2271 *Res. Part D*, 17, 332-340.
2272
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2274 **TABLE LEGENDS:**

2275

2276 **Table 1:** Contribution of non-exhaust sources to road transport and all primary sources of PM
2277 emissions in Europe and the U.S. These are according to current estimates from
2278 emission inventories for the respective regions, with comparisons done on a like-for-
2279 like basis, as far as is possible. Data are derived for the UK from NAEI (2020), for
2280 EU28 from EEA (2021), for California from CARB (2016) and CARB (2017) and
2281 for the USA from USEPA (2020a).

2282

2283 **Table 2:** Metal profiles of various NEE and mineral dusts based on selected ambient, tunnel
2284 and road dust studies. *Metals detected at a subset of sites in parenthesis.

2285 **Table 3:** Summary of quantitative mass apportionment studies of NEE in polluted urban
2286 environments. Notes: (a) Kerbside increment rather than overall concentration
2287 reported; (b) Downtown site; (c) Highway site.

2288

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2290 **FIGURE LEGENDS:**

2291

2292 **Figure 1:** Total exhaust and non-exhaust road transport emissions of PM_{2.5} from EU28
2293 countries from emission inventories reported by Parties under the Convention on
2294 Long-range Transboundary Air Pollution in 2020, covering years up to 2018 (EEA,
2295 2021).

2296

2297 **Figure 2:** Percentage share of total non-exhaust emissions of PM_{2.5} and PM₁₀ for EU28
2298 countries combined, as a percentage share of total road transport and of all national
2299 primary emissions (shown in the figure marked 'Total'). Based on data from EEA
2300 (2021).

2301

2302 **Figure 3:** UK emissions of NMVOCs from all road transport by sources according to the
2303 United Kingdom National Atmospheric Emissions Inventory, summary data
2304 available at <https://naei.beis.gov.uk>. Methodology for the construction of the VOC
2305 element of the NAEI is described in Passant et al., 2002.

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