AIR QUALITY EXPERT GROUP

Non-Exhaust Emissions from Road Traffic

Prepared for:

Department for Environment, Food and Rural Affairs;
Scottish Government; Welsh Government; and
Department of the Environment in Northern Ireland

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This is a report from the Air Quality Expert Group to the Department for Environment, Food and Rural Affairs; Scottish Government; Welsh Government; and Department of the Environment in Northern Ireland, on non-exhaust emissions from road traffic. The information contained within this report represents a review of the understanding and evidence available at the time of writing.

Front cover image credit: Mohamed Ghalaieny

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Contents

Executive Summary

Non-exhaust emissions (NEE) from road traffic refers to particles released into the air from brake wear, tyre wear, road surface wear and resuspension of road dust during on-road vehicle usage. These emissions arise regardless of the type of vehicle and its mode of power, and contribute to the total ambient particulate matter burden associated with human ill-heath and premature mortality. No legislation is currently in place specifically to limit or reduce NEE particles, so whilst legislation has been effective at driving down emissions of particles from the exhausts of internal-combustion-engine vehicles, the NEE proportion of road traffic emissions has increased. Data from the UK National Atmospheric Emissions Inventory indicate that particles from brake wear, tyre wear and road surface wear currently constitute 60% and 73% (by mass), respectively, of primary $PM_{2.5}$ and PM_{10} emissions from road transport, and will become more dominant in the future. Currently they contribute 7.4% and 8.5% of all UK primary $PM_{2.5}$ and PM_{10} emissions. Therefore to achieve further gains in $PM_{2.5}$ and PM_{10} air quality in relation to road transport sources requires attention to reducing non-exhaust emissions, not solely a focus on lowering exhaust emissions.

The magnitudes of non-exhaust emissions are, however, highly uncertain, particularly when compared to data for exhaust emissions. Emissions vary widely according to brake, tyre and road-surface material, and with driving style. The NEE emission factors used in inventories have a wide span of uncertainty – greater than a factor of two is typical – including uncertainty in splits between PM_{10} and $PM_{2.5}$ size fractions. The emission factors are also largely based on data from the 1990s and have not changed as vehicle designs and fleet composition have changed, in contrast to the regularly updated factors used for exhaust emissions.

The available data indicate that brake, tyre and road-surface wear contribute approximately equally to UK sources of NEE, and are predominantly derived from cars because of the much greater vehicle-km travelled for this class of vehicle. NEE particles are also an important source of metals to the atmosphere; the national inventory estimates NEE contributions of 47% and 21% for Cu and Zn, primarily associated with brake and tyre wear, respectively. The national inventory does not include estimates of road dust resuspension.

NEE are especially important in urban environments. The national inventory indicates that half of NEE occurs on urban roads, owing to the greater braking per km than on non-urban roads. Emissions may also be high in areas such as trunk-road exits. Tyre-wear emissions are estimated to be greatest on high-traffic trunk roads and motorways (both urban and rural).

Considerable measurement evidence shows NEE increase concentrations of PM_{10} and $PM_{2.5}$ and some metals at roadside although precise quantification of the NEE contribution is difficult. Data from London Marylebone Road indicate an NEE contribution (including resuspension) of 4-5 $\mu g\ m^{-3}$ to the roadside increment in PM, mostly in the coarse particle fraction ($PM_{10-2.5}$). Other studies, including dispersion modelling, also indicate total NEE contributions, including resuspension, of up to several $\mu g\ m^{-3}$ of PM_{10} at busy roadsides, and in the region 1-2 $\mu g\ m^{-3}$ for urban background in central London.

The most effective mitigation strategies for NEE are to reduce the overall volume of traffic, lower the speed where traffic is free-flowing (e.g. trunk roads and motorways), and promote driving behaviour that reduces braking and higher-speed cornering. Resuspension of particles from the road surface can be lowered by reducing the material that is tracked onto public road surfaces by vehicle movements in and out of construction, waste-management and similar sites; and potentially by road sweeping, street washing and application of dust suppressants to street surfaces, although the impacts on airborne PM from trials of these latter approaches have so far proven inconsistent and any benefits have been short-lived.

Regenerative braking does not rely on frictional wear of brake materials so vehicles using regenerative braking totally or partially, for example electric vehicles, should have lower brake wear emissions. However, tyre and road wear emissions increase with vehicle mass, which has implications for any vehicle with a powertrain that is heavier (for example due to additional battery and hardware mass) than the equivalent internal-combustion-engine vehicle it replaces. The net balance between reductions in brake wear emissions and potential increases in tyre and road wear emissions and resuspension for vehicles with regenerative braking remains unquantified, and will depend upon road type and driving mode, as both influence the balance between the different sources of emissions. In locations where brake wear makes a major contribution to overall NEE, it seems likely that there will be a net benefit, but this has yet to be demonstrated. Other as yet unproven technological mitigation methods include trapping brake wear particles prior to emission, and mandating formulation of low-wear/low-emission tyres, brake pads and road surfaces.

AQEG recommends as an immediate priority that NEE are recognised as a source of ambient concentrations of airborne PM, even for vehicles with zero exhaust emissions of particles.

A further priority is to work towards a consistent approach internationally for measurement of NEE and to update and narrow the uncertainties in their emission factors. Such a programme

of work could form the basis for subsequently including criteria on brake and tyre wear emissions in future type approvals and regulations governing formulation.

AQEG also recommends that further studies be conducted to quantify the efficacy of technical solutions on NEE reductions; in particular, to understand gains from use of regenerative braking versus potential increased tyre and road wear due to additional mass of vehicles incorporating such braking.

1 Introduction

Road traffic has long been recognised as a major source of air pollution due to emissions of a range of gaseous pollutants, most notably carbon monoxide, oxides of nitrogen and volatile organic compounds, as well as particulate matter. The gaseous pollutants are present in exhaust emissions, and, until recently, the dominant source of particles was also from the vehicle exhaust.

However, road transport is also an important source of 'non-exhaust emissions' (NEE) of particles, which are produced from frictional processes associated with vehicle usage: predominantly from brakes, tyres and the road surface. Whilst regulations set by the European Union have led to progressive reductions in the emissions of the regulated gaseous pollutants and of particulate matter from the exhausts of new vehicles, the non-exhaust emissions are not currently targeted by emissions regulations. Therefore, as the exhaust emissions have fallen, the proportion of non-exhaust emissions to the total emissions from road traffic has increased. Data from the UK National Atmospheric Emissions Inventory (NAEI) indicate that emissions of non-exhaust particles from road transport already exceed those from the exhaust, and their proportion is projected to increase in the future. Therefore, to achieve further improvements to $PM_{2.5}$ and PM_{10} air quality relating to road transport sources requires attention to reducing NEE, and not solely on approaches focused on lowering exhaust emissions (See also the text in Box 1 on 'zero emission' vehicles in this regard.)

However, quantitative data on the magnitude of non-exhaust emissions are sparse and highly uncertain, particularly when compared to data for exhaust emissions. The exact contribution of non-exhaust emissions to air quality locally and nationally is therefore currently subject to considerable certainty.

The aim of this AQEG report is to summarise current evidence for the non-exhaust emissions of particles from road transport at the point of on-road usage. The equivalent emissions from off-road vehicles are not included. Railway transport is also a source of non-exhaust particle emissions but there are currently no requirements to include these emissions in national inventories. Some European countries, but not the UK, provide some information on railway non-exhaust emissions and some information on railway NEE is presented in an Appendix to this report.

Box 1: Zero emission vehicles

The reductions in road transport exhaust emissions, and in particular the increasing market in electric vehicles, has bolstered use of the terminology 'zero emission vehicle'. However, non-exhaust vehicle emissions arise irrespective of the powertrain (conventional fuel, electric, fuel-cell, hydrogen, etc.). Some designs of electric buses also incorporate diesel powered heating systems which will be an additional source of emissions, as are diesel-powered refrigeration units on goods vehicles.

There may also be air pollutant emissions associated with displacement of emissions from the vehicle itself to somewhere further up the energy-supply chain, for example at an electricity generating facility, depending on the source of the electricity.

The terminology zero emission vehicle can therefore be misleading. Usage of the terminology 'zero exhaust emission vehicle' is more precise and is preferred. See also https://uk-air.defra.gov.uk/library/aqeg/zero-emission-vehicles

1.1 What is non-exhaust emissions particulate matter?

Non-exhaust particles arise from a range of vehicle-related sources. The main contributors are the following:

a) **Brake wear.** Standard frictional brakes on a vehicle function by virtue of the friction between a brake pad and a rotating disc or drum when the two are forced together by application of pressure to the braking system. The frictional process causes abrasion both of the brake pad and of the surface of the disc or drum leading to the release of particles, a substantial fraction of which become airborne.

b) **Tyre wear.** The surface of a tyre when in contact with the road is steadily abraded by contact with the road surface. This leads to release of large quantities of small rubber particles which cover a wide range of sizes. The larger particles will typically remain on the road surface until washed off in drainage water. However, the size range extends into sizes below 10 micrometres diameter and hence contributes to PM_{10} (and to $PM_{2.5}$). The smaller abraded particles are liable to become airborne contributing to non-exhaust particles in the atmosphere. If rubber tyre wear particles are considered to be a form of 'microplastics' then tyre wear would constitute an important source of microplastics into the environment, both via the airborne route but also via wash-off of the coarser tyre abrasion material remaining on the road surface – see Box 2 for further discussion of this. In this report, the term tyre wear particles is used without any implication as to whether they are also considered microplastic particles.

c) **Road surface wear.** The friction between the tyre surface and the road surface which leads to tyre abrasion is also liable to abrade the road surface, especially where this is already fragmenting. Hence, road surface wear particles are also released to the atmosphere. Some studies have suggested that road wear particles are internally mixed with tyre rubber in the particles generated through this abrasion process (see again also Box 2).

d) **Resuspended road dust.** Dusts from a number of sources accumulate on road surfaces. These originate from dry and wet deposition of airborne particles, especially coarser particles such as those deriving from soil. Additionally, abrasion products from the vehicle may deposit on the road contributing to the road surface dusts. Some of this material is in the PM_{10} size range when depositing to the road surface and the action of tyres on surface dusts may also cause some grinding leading to the creation of smaller particles from the coarser dusts. Studies of road surface dusts have shown a substantial fraction to be within the $PM_{2.5}$ and PM_{10} size ranges. Such particles are rather easily suspended from the road surface, both by shear forces at the tyre-road interface and by atmospheric turbulence in the wake of the vehicle. There is also evidence that elevated wind speeds contribute to the resuspension of surface dusts.

In addition to these major contributors, there are also other abrasion sources associated with the vehicle such as wear of exposed drive belts, rubber gaiters and clutch plates, although in the latter case the majority of the abrasion products are contained by the clutch housing.

The operation of disc brakes and drum brakes relies on friction between brake pads or brake shoes against the disc or drum respectively. The wear of the components will typically produce relatively coarse airborne particles, but the high temperatures associated with brake components will typically promote the generation of ultrafine particles. Whilst many different materials have been and are being used for these components, most researchers have reported Fe, Cu, Zn and Pb to be the most abundant metals in the brake lining, with the Pb component declining rapidly in recent years. The metals Ba and Sb are also reported to be tracers of brake wear and are less susceptible to also having contribution from other sources. Metals are also present in tyre wear particles, with Zn and Cd most notable. In the near-road environment, non-exhaust emissions contribute a major source of a number of these metals into ambient air, particularly Cu and Zn.

Box 2: Non-exhaust emissions and microplastics

The extent to which NEE contributes to the microscopic plastic particles (microplastics) entering the environment depends in part on the definition of plastic. There is some disagreement on which polymers are “plastics”. As discussed in Hartman et al. (2019), the ISO 472: 2013 definition of plastic is “material which contains as an essential ingredient a high molecular weight polymer and which, at some stage in its processing into finished products, can be shaped by flow”. Some elastomers (e.g. rubbers) are excluded from this definition of plastic. This definition however reflects the historic industrial landscape rather than perspectives about the behaviour of rubber fragments in the environment. A further consideration in terming tyre wear as plastic arises from the way in which tyre wear particles contain road wear fragments too, as shown in Figure 1. Kreider et al. (2010) and Panko et al. (2013) estimated that tyre wear particles comprised around 50% tyre tread and around 50% road surface.

Figure 1: Scanning electron microscope photo of tyre and road wear particles with characteristic morphology of tread rubber and mineral incrustations from pavement. Reproduced with permission from Panko et al. (2019).

Others such as Kole et al. (2017) have included rubber within their definition of plastics. If included, rubber production would add 27 million tonnes per year to the annual global production of plastics of around 211 million tonnes. This does not mean that all this material enters the environment. Understanding the environmental pathways is a challenge but, if defined as plastic, then tyre wear could be adding 5-10% to the global total of microplastics entering the oceans each year (Kole et al. 2017). Another estimate from the International Union for Conservation of Nature and Natural Resources (Boucher and Friot, 2017) is that erosion of tyres whilst driving contributes 28% of the releases of primary microplastics to the world’s oceans. These estimates make wear and tear from tyres at least as important as plastic bottles, bags and fibres released from clothing during washing. The wear of thermoplastic road markings might also be included within the definition of plastics adding further to the contribution of NEE to environmental microplastics.

1.2 Why are road traffic non-exhaust emissions important?

Non-exhaust emissions from road traffic contribute to airborne concentrations of both fine and coarse particles and hence to $PM_{2.5}$ and PM_{10}. The estimates from the National Atmospheric Emissions Inventory outlined in Chapter 2 indicate that the emissions from brake wear, tyre wear and road surface wear collectively now exceed those from the exhaust of the UK vehicle fleet. The Committee on the Medical Effects of Air Pollutants (COMEAP) has estimated that exposure of the UK population to particulate air pollution contributes to an effect equivalent to around 29,000 deaths across the country annually (COMEAP, 2010). COMEAP has also examined the evidence for variations in toxicity between particles of different chemical composition or from different sources and has concluded that present evidence is insufficient to judge whether particles of particular composition or from particular sources have higher toxicity (COMEAP, 2015). This means that COMEAP is unable to recommend differential coefficients for quantification of health effects, and continues to recommend that concentration-response coefficients linking mortality with $PM_{2.5}$ mass concentration be applied to all particles within the size range. Consequently, on the basis of current emissions inventory estimates and toxicity evidence, non-exhaust particles from the UK road traffic fleet should be considered as potentially having a greater public health impact than the exhaust particles.

Air quality policy within the European Union generally, and in the UK specifically, has focussed upon reducing public exposure to harmful air pollutants. This has included reducing exposure to airborne particulate matter and there have been some notable successes. The sources of particulate matter with well-defined and constrained sources such as those from vehicle exhaust and industrial processes have been subject to steadily tightening emissions standards and there are very limited opportunities of further reductions without incurring substantial cost. Consequently, attention is now focussing upon those sources which are less well controlled such as domestic wood burning and those which are not subject to control such as non-exhaust emissions from road traffic.

1.3 Vehicle mass and non-exhaust emissions

Non-exhaust emissions have very different origins and characteristics compared with gaseous or particulate exhaust emissions. While a detailed understanding of the processes leading to NEE is highly complex, it is possible to develop a broad understanding of the important issues. One important underlying factor that has a direct influence on NEE is vehicle mass, since

mass influences the amount of friction with the road surface and the energy dissipated through braking, both of which are sources of NEE.

The friction of a vehicle against the road is important for both tyre wear and resuspension of particles from the road surface. The frictional force at the surface = $C_r.M.g$, where M is the vehicle mass, C_r the coefficient of rolling resistance, and g the gravitational acceleration constant. Values of C_r depend on the surface but tend to be higher for 'rougher' surfaces. The pressure of a tyre also has an influence on the rolling resistance with lower pressure tyres having a higher rolling resistance with the surface.

With regard to the brake wear source, when a vehicle brakes the kinetic energy of the vehicle is dissipated through the braking system and, in a conventional frictional braking system, is lost as heat. A vehicle's kinetic energy is proportional to its mass. For an example car of 1,500 kg travelling at 70 mph and braking to a stop, an energy of 735 kJ must be dissipated through the braking system (ignoring any losses to rolling resistance or aerodynamic drag). Higher mass vehicles require higher levels of energy dissipation, larger braking systems and consequently increased wear. In contrast, in a regenerative braking system much of the vehicle's kinetic energy is channelled into on-vehicle energy storage, typically a battery.

The generation of heat on brake pads and discs is also important in its own right. For example, high temperatures have been shown to promote the generation of UFP (Perricone et al., 2018). About 90% of the braking heat energy goes into the disc not the pad because of the greater mass and thermal conductivity of the former. In the example above, the energy lost to the braking system would be sufficient to raise the temperature of 10 kg of steel from about 15 °C to 117 °C, based on typical specific heat capacities of steel. Under real driving conditions there can be situations that lead to repeated heating of brake discs and pads in quick succession, e.g. braking down a long hill. Under these conditions, temperatures could rise further but would also be offset by cooling losses.

The above considerations help point to where NEE might be most important. Tyre wear and resuspension of particles at the surface are both likely dependent on vehicle mass and would be expected to be of importance under many conditions – although dependent on the characteristics of road surfaces and other environmental factors. By contrast, brake wear emissions (for conventional frictional braking systems), whilst also dependent on vehicle mass, would be expected to be much more spatially heterogeneous e.g. of most importance close to junctions or on steep downhill gradients.

2 Sources and Characteristics of Non-Exhaust Emissions of PM

Emission factors are available for tyre wear, brake wear and road surface wear which have been measured under controlled laboratory conditions. Currently, there are no standardised test protocols, although one is currently under development for brake wear particles (see Section 2.3). Considerable uncertainties attach to these emission factors for reasons including the following:

- Variability in materials. Brake pads are of highly variable composition, and there are differences, but of smaller magnitude, in brake discs and tyre rubber compound. Road surface materials vary widely in composition and texture, as well as in their state of repair. Consequently, even when subject to the same external forces, different brakes or road surfaces will generate particles with variable efficiency leading to differences in emission factors which are hard to capture in detail as it is not feasible to test across the whole range of properties of the materials.

- For practical reasons, emission factors have to be related to road type and/or average speed whereas in practice there are other factors such as congestion or road gradient which influence the extent to which brakes and tyres are abraded. This leads to substantial ranges of emission factors when expressed per kilometre of travel.

Substantial differences in emission factors for tyre wear, brake wear and road surface wear are to be expected for different road types. Hence, freely flowing high speed traffic will generate very low levels of brake wear particles, but is liable to create a larger mass emission of tyre wear and road surface wear particles. On the other hand, a congested highway with frequent stopping and starting is liable to generate far larger brake wear emissions (see Section 2.2).

Further uncertainties relate to electric vehicles. Battery electric vehicles can use either regenerative braking or conventional friction brakes. Regenerative braking involves cutting power to the electric motor which then continues to rotate due to the inertia of the vehicle and acts as a generator recharging the batteries. In doing so, it is subject to a substantial reverse force which slows the vehicle. Unlike the friction brakes, this does not lead to significant generation of particles, but the ratio of regenerative to friction braking will depend upon driving style and road conditions and is hence difficult to predict.

An additional source of non-exhaust vehicle-related particles into the air is resuspension of road dust. However, research on resuspension emissions is highly incomplete and estimates from this source do not need to be included in European national inventories, nor are there currently any recommended guidelines for estimating national emissions from this source.

The algorithm used to predict resuspension of road dust from paved roads by the US Environmental Protection Agency AP-42 Compilation of Air Pollutant Emission Factors is the following,

$E = k\ (sL)^p\ W^b$

where *sL* is silt loading, *W* is vehicle mass and p = 0.91 and b = 1.02. In this case the silt loading refers to the surface loading of particles less than 75 µm on the road surface and consequently this empirically determined relationship relates to a much wider range of particle sizes than the PM_{10} or $PM_{2.5}$ size range. This near-linear empirical relationship between resuspension emissions and vehicle mass is consistent with the linear relationship between road-surface frictional force and vehicle mass described in Section 1.3, and is relevant to predictions of how this source of NEE will change when internal-combustion vehicles in the current fleet are replaced by those with heavier power trains (which includes battery and hybrid vehicles).

From work conducted in Europe, Padoan et al. (2018) proposed the following alternative equation for road resuspension emissions,

$EF\ (\text{mg VKT}^{-1}) = a\ (MF10)^b$

where *MF10* is the 'mobile fraction' of road dust of diameter <10 µm (in mg m^{-2}), and *a* and *b* are empirically determined coefficients (VKT = vehicle km travelled). In contrast to the AP-42 approach, the emission estimate from this equation is for the PM_{10} size range. The value for *MF10* can either be measured directly from the road surface (the PM_{10} capable of resuspension in an airflow of 30 L min^{-1}) or predicted from an empirical relationship that incorporates a measure of the road surface texture, the traffic intensity, and the distance from the closest braking zone (Padoan et al., 2018).

Venkatram (2000) criticised the USEPA model on the grounds that it can yield highly uncertain emission estimates because it lacks a mechanistic basis; its formulation is highly dependent on the dataset used to derive it, and the accuracy of the model is completely determined by the methods used to measure emissions. Padoan et al. (2018) provide some evidence of

testing the fit to their model, but it has yet to be more widely evaluated and currently such predictive methods are open to large uncertainties.

It is notable that the two equations differ in that the USEPA method includes vehicle mass whilst the other does not, and neither considers the aerodynamics of individual vehicles which may affect the resuspension of particles in the turbulent wake of the vehicle. In addition, neither equation takes any account of vehicle speed which might also be expected to influence the resuspension process, nor do they incorporate current and recent weather conditions which dramatically alter surface dust amount and mobility. Thus, as with all empirical models, these models for road resuspension may fail severely when used predictively for conditions outside of those used to derive them.

2.1 UK National Atmospheric Emission Inventory estimates of NEE

National emission inventories cover non-exhaust sources of PM from tyre and brake wear and road surface wear (road abrasion). These inventories are reported by countries to the EU under the National Emissions Ceilings Directive (NECD) and the United Nations Economic Commission for Europe (UNECE) Convention on Long-Range Transboundary Air Pollution (CLRTAP). The revised NECD (2016/2284/EU), which entered into force on 31 December 2016, sets new emission reduction commitments (ERCs) for each Member State for the total emissions of $PM_{2.5}$ (and other pollutants) in 2020 and 2030 (NECD, 2016). Inventories covered under the Directive must include these NEE sources. Similar emission reduction commitments for $PM_{2.5}$ are required under the CLRTAP for 2020.

As highlighted above, although road dust resuspension is recognised as an important source of PM in ambient air, national inventories are not required to report estimates of these emissions, which depend on many local factors and are hard to predict.

2.1.1 Inventory approach for PM_{10} and $PM_{2.5}$

Most countries follow the methodology for estimating emissions from tyre and brake wear and road surface wear given in the 2016 version of the EMEP/EEA Air Pollutant Emissions Inventory Guidebook (EMEP/EEA, 2016). This provides a fairly simple approach which combines PM emission factors in milligrammes emitted per kilometre (mg/km) for passenger cars, light goods vehicles, heavy duty vehicles (HGVs and buses) and two-wheelers, with

vehicle kilometres travelled per year. The method and emission factors in the Guidebook have not been updated for nearly 15 years and are based on the information available at the time, mostly on wear rates, and a number of assumptions. More information on the Guidebook method can be found at http://www.eng.auth.gr/mech0/lat/PM10/.

Some countries have used emission factors based on their own literature search (e.g. the Netherlands) or have used evidence from country-specific information and research. These are generally based on the total mass loss of tyre or brake material resulting from the wear process and estimates on the amount that remain airborne in the PM_{10} and $PM_{2.5}$ range. The Scandinavian countries have been particularly active in this area and, for example, have taken account of the effect of studded tyres resulting in higher emissions from road wear (e.g. see Sweden's Informative Inventory Report 2014, https://www.naturvardsverket.se/upload/sa-mar-miljon/klimat-och-luft/luft/luftfororeningar/iir-sweden-2014.pdf).

The UK's National Atmospheric Emissions Inventory (NAEI) for tyre and brake wear and road abrasion uses the Tier 2 inventory method and emission factors in the EMEP/EEA Emissions Inventory Guidebook (NAEI, 2018)[1]. This approach provides mg/km emission factors for Total Suspended Particulates (TSP) for passenger cars, LGVs, HDVs and two-wheeled vehicles, together with PM_{10} and $PM_{2.5}$ mass fractions to combine with the TSP factors. The TSP factors for tyre and brake wear are used with an average speed correction factor which implies higher emission factors at lower speeds, on the basis of greater braking and cornering per km at lower speeds. For heavy duty vehicles, a further correction factor is applied to take account of the load carried by the truck and in the case of tyre wear on the number of wheel axles. No such speed and load correction factors are provided for road surface wear emissions.

*Table **1*** shows the average PM_{10} emission factors for tyre and brake wear for vehicles in the UK at typical urban, rural and motorway speeds. These were derived in the NAEI from the factors in the EMEP/EEA Guidebook. *Table **2*** shows the average PM_{10} emission factors for road abrasion taken from the Guidebook for all road types and speeds.

[1] The EMEP/EEA Guidebook for compiling emission inventories provides different approaches according to the availability of activity data. The simplest approach is referred to a Tier 1 approach and is recommended when a country has minimal information available on source activities to make an estimate of emissions. Tier 2 or Tier 3 involve more detailed approaches when appropriate activity data are available. The Tier 2 approach is the most detailed approach in the guidebook for estimating non-exhaust emissions. A Tier 3 approach is available in the Guidebook for estimating exhaust emissions and is used in the UK's national inventory.

Table 1: Emission factors for PM_{10} from tyre and brake wear.

mg PM_{10} / km		Tyre	Brake
Cars	Urban	8.7	11.7
	Rural	6.8	5.5
	Motorway	5.8	1.4
LGVs	Urban	13.8	18.2
	Rural	10.7	8.6
	Motorway	9.2	2.1
Rigid HGVs	Urban	20.7	51.0
	Rural	17.4	27.1
	Motorway	14.0	8.4
Artic HGVs	Urban	47.1	51.0
	Rural	38.2	27.1
	Motorway	31.5	8.4
Buses	Urban	21.2	53.6
	Rural	17.4	27.1
	Motorway	14.0	8.4
Motorcycles	Urban	3.7	5.8
	Rural	2.9	2.8
	Motorway	2.5	0.7

Table 2: Emission factors for PM_{10} from road abrasion.

mg PM_{10} / km	Road abrasion
Cars	7.5
LGVs	7.5
HGVs	38.0
Buses	38.0
Motorcycles	3.0

There are considerable uncertainties in the wear rates on which these PM_{10} emission factors are based. The emission factors in the 2016 version of the EMEP/EEA Emissions Inventory Guidebook and the methods used to correct them for different vehicles, speeds and loads are taken from a review undertaken for the UNECE Task Force on Emission Inventories and Projections (TFEIP) supporting the development of the Guidebook. This review is available

at https://www.eng.auth.gr/mech0/lat/PM10/ and provides a list of the literature sources used in their derivation.

For tyre wear emissions from passenger cars, the TFEIP source indicates that PM_{10} emission factors come from seven literature sources between 1997 and 2002 providing estimates of wear rates and two sources in the 1990s, including a USEPA source, providing direct estimates of PM_{10} emissions. The wear rates varied from 40-97 mg/km and were used in conjunction with an assumption that 10% of tyre wear material is suspended as PM in the 10 micron range, a fraction which is said to be at the upper end of the range in the literature. The average PM_{10} emission factor is quoted as 6.4 mg/km but with a range of 4.0 to 9.7 mg/km. In a report for the Joint Research Centre (JRC), Kouridis et al. (2010), from the same team responsible for the Guidebook emission factors, quote a standard deviation in the tyre wear emission factors for a passenger car of 0.8 mg/km. To put this in context, a fleet-average PM exhaust emission factor for a Euro 5 diesel car from the same Guidebook source is given as 2.1 mg/km; a range is not given and the Guidebook only gives qualitative statements on the uncertainties in the exhaust emission factors, though the NAEI has made a very rough estimate of ±40% uncertainty in PM exhaust emission factors for diesel cars based on the degree of scatter and variability of some raw data from tests done in the UK.

In the case of heavy duty vehicles, the TFEIP source indicates that PM_{10} emission factors come from three literature sources in 1997 providing estimates of wear rates and one source in 1999 providing direct estimates of PM_{10} emissions. Tyre wear emissions from heavy duty vehicles are characterised by the variability in the number of axles and by the wide range of a truck's load. Therefore, the number of axles and the load factor need to be taken under consideration for the calculation of HDV emissions from tyre wear. The uncertainty range of PM_{10} emission factors given in the TFEIP source is 14-54 mg/km.

For brake wear emissions from passenger cars, the TFEIP source indicates that PM_{10} emission factors come from four literature sources between 1999 and 2002 providing estimates of wear rates. The wear rates varied from 9-20 mg/km and were used in conjunction with an assumption that ~50% of brake wear material is suspended as PM in the 10 micron range, as proposed by USEPA (1995) and TNO (1997). The average PM_{10} emission factor is quoted as 7.5 mg/km but with a range of 4.4 to 10 mg/km. Kouridis et al. (2010), from the same team responsible for the Guidebook emission factors, quote a standard deviation in the brake wear emission factors for a passenger car of 0.8 mg/km.

In the case of heavy duty vehicles, the TFEIP source indicates that PM_{10} emission factors for brake wear come from two literature sources in 1999-2001 providing estimates of wear rates and one sources in 1998 providing direct estimates of PM_{10} emissions. Brake wear emissions from heavy duty vehicles are characterised by the wide range of a truck's load. Therefore, the load factor needs to be taken under consideration for the calculation of HDV emissions from brake wear. The uncertainty range of PM_{10} emission factors given in the TFEIP source is 23-42 mg/km.

What is clearly apparent is that all the emission factors for these non-exhaust sources come from the same era and have not been updated in the EMEP/EEA Inventory Guidebook in over 15 years, yet they are still used in national inventories by most countries in Europe, including the UK. These factors were based on analysis of data available at the time the review for the TFEIP was undertaken. Whilst the range in emission factors, and indeed the uncertainty analysis carried out by Kouridis et al. (2010), may reflect the variability in measurements undertaken in that era, they may not be a true reflection of the uncertainties in emission factors representing current vehicles, tyre and brake materials and the Guidebook factors could be systematically biased in one direction or another. Changes in tyre and brake materials, vehicle design and braking technologies could mean that current emission factors are outside the ranges indicated above. There is an urgent need for further direct measurements of emission factors for current vehicles and technologies to test this and update the factors for use in emission inventories.

Emission factors for road abrasion are highly uncertain, but whilst factors are provided in the Guidebook from Klimont et al. (2002), no estimates of their uncertainties are given. This source is also expected to be affected by changes in tyre materials and road surfaces.

Another source of uncertainty is the fraction assumed in the $PM_{2.5}$ range. Most countries, including the UK, France and Germany use the $PM_{2.5}/PM_{10}$ ratios shown in *Table **3*** for tyre wear, brake wear and road abrasion emissions, taken from the EMEP/EEA Guidebook. However, not all countries use these ratios. The Netherlands, for example, assumes a ratio of 0.2 for tyre wear and 0.15 for brake wear and road abrasion implying a much larger share of PM emissions occur in the coarse fraction. Sweden uses a ratio of 0.2 for all these sources while Finland uses a ratio of 0.09 for road abrasion which is mainly due to the increased use of studded tyres leading to a higher proportion of emissions in the coarse range. The inventory reports for Sweden and Finland use a correction factor of 50 for PM_{10} road abrasion emissions from studded tyres relative to non-studded tyres.

Table 3: Fraction of PM_{10} emitted as $PM_{2.5}$ for non-exhaust traffic emission sources.

	$PM_{2.5}/PM_{10}$
Tyre wear	0.7
Brake wear	0.4
Road abrasion	0.54

No countries take account of any change in emission factors for NEE over time due to changes in vehicles and technologies. The consequence of this is that overall non-exhaust emissions continue to increase over time with increases in numbers of vehicles and kilometres travelled, in contrast to exhaust emissions of PM which all national inventories show are decreasing with time as newer vehicles meeting tighter emission standards enter the fleet, with increasing number of diesel vehicles fitted with particulate filters. Emission factors for NEE sources of PM have not been developed for inventories accounting for factors that might affect emissions such as vehicle mass, different tyre materials and braking systems and alternative powertrains such as hybrid and battery electric vehicles with regenerative braking systems. Some of these are considered in Chapter 6. Ricardo Energy & Environment carried out a review for the German auto industry (Verband der Automobilindustrie) on the contribution of brake wear emissions to particulate matter in ambient air (VDA, 2017). This gave an overview of current and developing brake wear system technologies and considered opportunities to reduce particle emissions from brakes and the vehicle segments to which they apply. The review gave a semi-quantitative assessment of the potential impact of eight different braking technologies, the most beneficial in the short term considered to be regenerative braking applied to light duty vehicles and buses. See also Chapter 5 for further discussion of NEE abatement approaches.

2.1.2 What does the inventory indicate about the trend in NEE in the UK?

Using the Guidebook emission factors and vehicle activity data, the NAEI reports the trends in UK tyre wear, brake wear and road abrasion emissions of PM_{10} and $PM_{2.5}$ shown in Figure 2 and Figure 3. For comparison, emissions from vehicle exhausts are also shown. The emissions shown from 2000-2016 are from the latest version of the reported UK inventory representing actual vehicle activities, while emissions from 2017-2030 are projections in

emissions based on DfT's traffic growth assumptions and in the case of exhaust emissions reflect the turnover in the vehicle fleet with the penetration of new vehicles meeting tighter Euro standards for PM emissions (Defra, 2018 and EIONET, 2018).

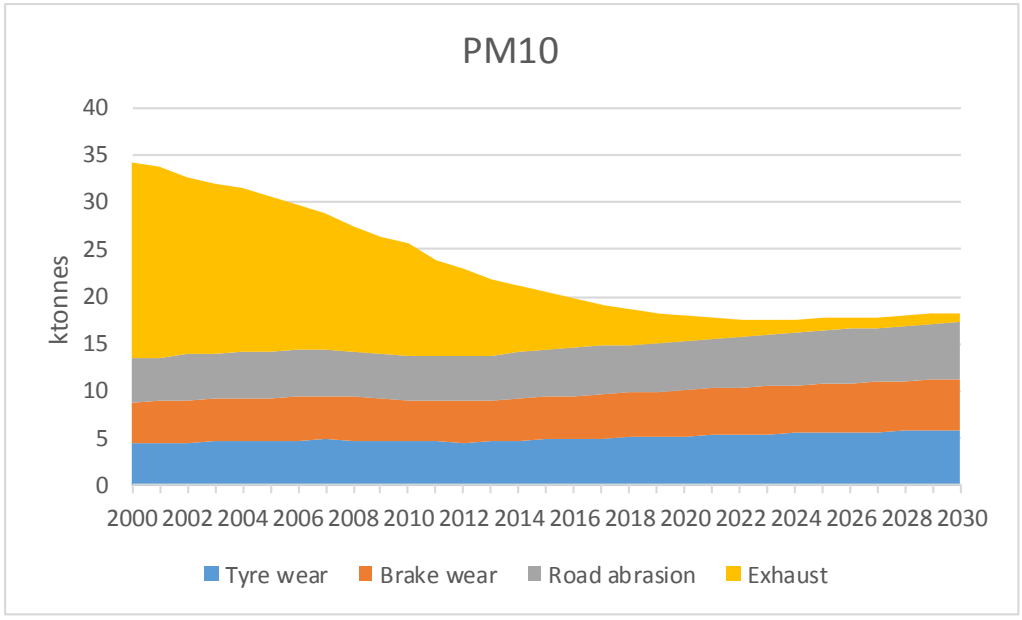

Figure 2: UK emissions of PM_{10} from road transport.

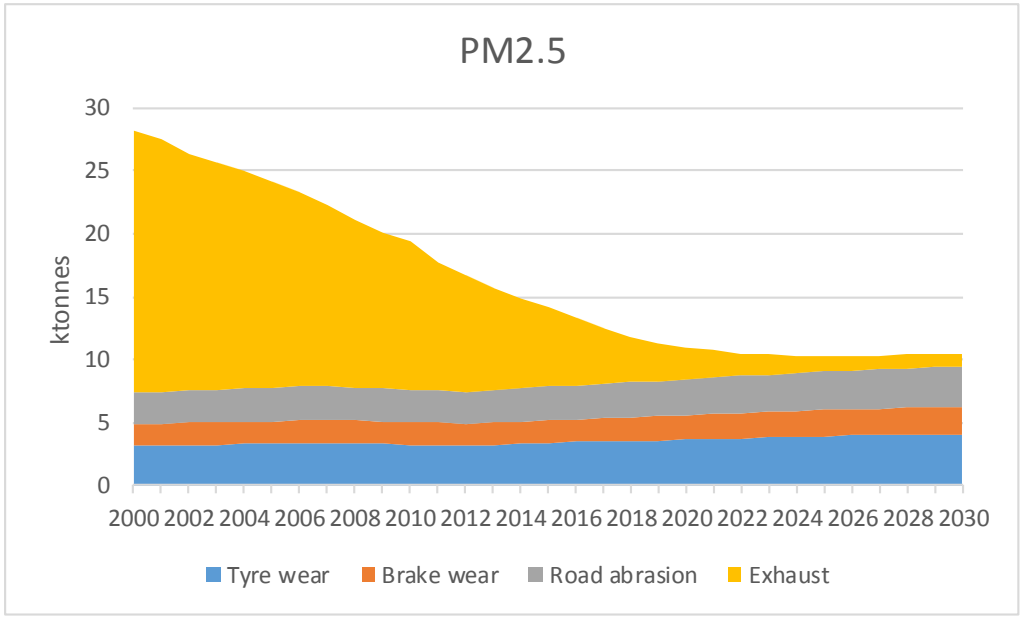

Figure 3: UK emissions of $PM_{2.5}$ from road transport.

These figures show how as vehicle exhaust emissions have declined, the non-exhaust emissions have been slowly increasing with increasing traffic levels and are becoming a much larger share of overall PM_{10} and $PM_{2.5}$ traffic emissions. The proportion of total NEE from brake wear, tyre wear, road surface wear has increased from 39% of total UK road transport emissions of PM_{10} in 2000 to 73% in 2016; for $PM_{2.5}$ the proportion of NEE has increased from 26% in 2000 to 60% in 2016[2].

Without any NEE abatement this trend is predicted to continue so that by 2030, the non-exhaust sources will contribute to 94% of total UK road transport emissions of PM_{10} and 90% of $PM_{2.5}$.

The projected increase in NEE to 2030 is based on the assumption that traffic will increase in future years relative to current levels. This is an assumption according to DfT's traffic forecasts. The NAEI emission projections shown in Figure 2 and Figure 3 are derived from DfT's 2015 traffic forecasts (RTF15 – Scenario 1). Figure 4 shows the trend in total urban UK traffic expressed as billion vehicle kilometres by vehicle type historically from 2002 to 2016 and forecast to 2035.

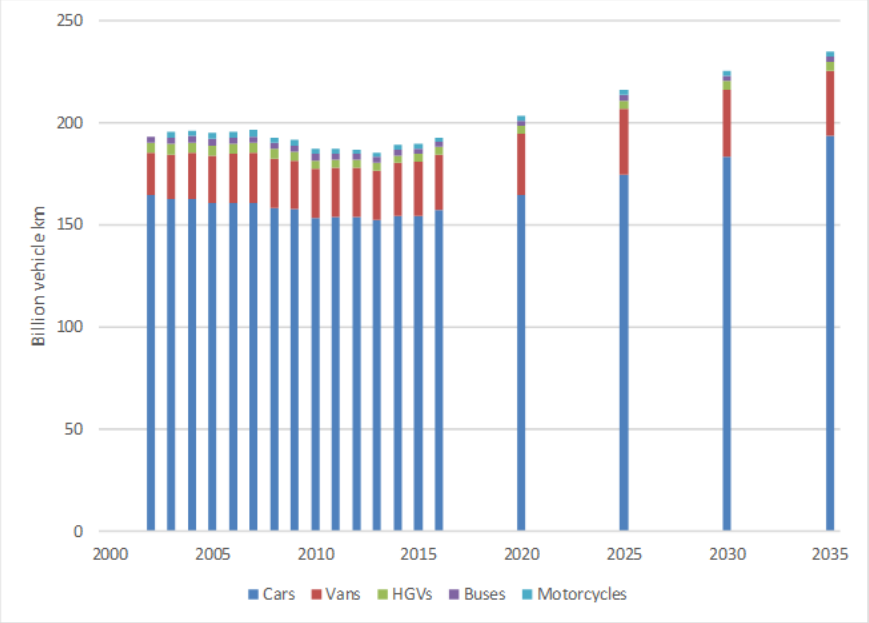

Figure 4: Urban UK vehicle kilometres. Historical data based on DfT traffic statistics; forecasts to 2035 are based on DfT traffic forecasts RTF15 – Scenario 1

[2] All UK inventory values in this report refer to emissions according to fuel used. The UK is required to report inventories to the NECD and CLRTAP on a fuel sold basis but can choose to also report on a fuel used basis. The UK's projections and NECD and CLRTAP emission reduction targets are on a fuel used basis.

This shows that there has actually been little overall change in urban traffic since 2002, largely due to the halt in growth which had been occurring prior to 2008, presumably due to the impact of the economic recession, though it can be seen that urban traffic levels have been on the rise again since 2013. More recent traffic statistics from DfT indicated that growth continued in 2017. It remains to be seen whether traffic growth will occur at the rate predicted by DfT, leading to the growth in NEE in urban areas currently predicted by the NAEI. Figures for total UK traffic (i.e. including rural traffic) show a similar trend, although with a slightly different vehicle mix with the higher contribution from HGV activity.

It is apparent from these figures that each of the three NEE sources contribute roughly similar amounts to the overall inventory and that there is no dominant source. This reflects the similar magnitudes of the emission factors shown in *Table **1*** and Table 2. Figure 5 demonstrates the emissions of PM_{10} from each of these three NEE sources in 2016 broken down by vehicle type. This chart shows the dominance of passenger cars to overall NEE, being responsible for 64% of all NEE emissions in 2016 due to the high activity levels (vehicle km) by these vehicles.

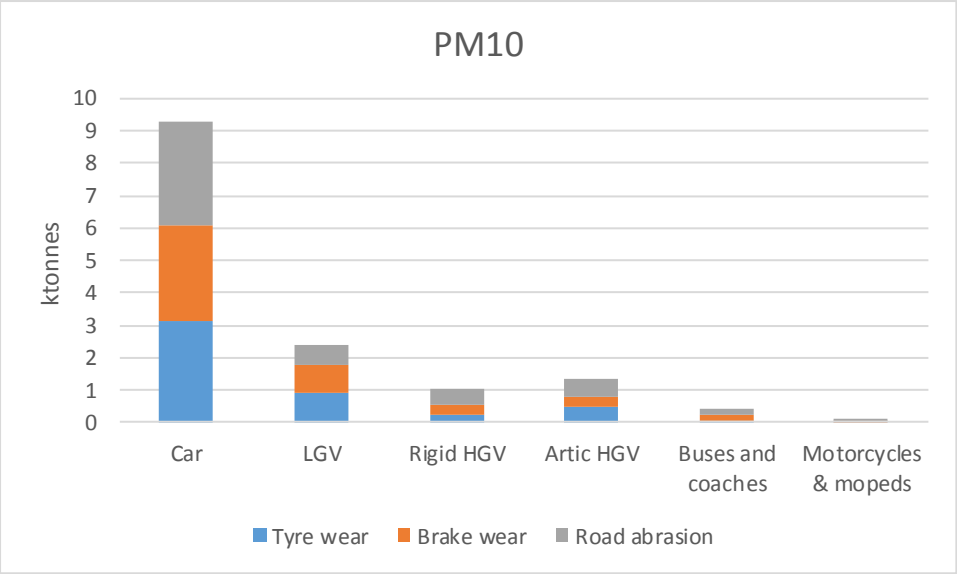

Figure 5: UK emissions of PM_{10} from road transport in 2016 by vehicle type

Another observation that can be made from the inventory is that almost half of the overall UK non-exhaust emissions from brake wear, tyre wear, road surface wear occur on urban roads. This is a reflection of the traffic levels by vehicle type on different road types according to DfT

traffic statistics and the higher emission factors shown in the Guidebook for tyre wear and brake wear under urban speed conditions.

The NAEI indicates that not only are non-exhaust sources of PM making a larger contribution to total UK traffic emissions, they are also making a growing contribution to the total UK inventory covering all primary emission sources. In 2000, NEE from brake wear, tyre wear, road surface wear were 5.8% of total UK PM_{10} emissions and 4.9% of total $PM_{2.5}$ emissions and this has increased to 8.5% of total UK PM_{10} emissions and 7.4% of total $PM_{2.5}$ emissions in 2016. By 2030, it is predicted to rise to 9.5% of total UK $PM_{2.5}$ emissions if no abatement measures on NEE are introduced.

2.1.3 Trends in non-exhaust emissions of PM in other European Countries

The trends in UK NEE can be compared with trends according to inventories reported by other European countries[3]. Figure 6 shows emissions of PM_{10} from non-exhaust traffic sources from 2000-2016 for Germany, France, Netherlands, Sweden, Denmark, Finland and the UK. Figure 7 shows the corresponding trend in emissions of $PM_{2.5}$. The trends are broadly similar for all countries showing a slow upward trend in line with increases in traffic. The inventory trends for Germany and France are quite similar to that of the UK. One notable difference is for Sweden and Finland which show a much larger amount of PM_{10} emitted relative to $PM_{2.5}$ across the time-series compared with other countries. This is mainly due to these countries accounting for the effect of studded tyres on road abrasion emissions which fall mainly in the PM_{10} range.

Figure 8 compares the breakdown in emissions from tyre and brake wear, road abrasion and exhaust emissions of PM_{10} and $PM_{2.5}$ for 2000, 2005, 2010 and 2016 for a number of countries. In these plots, emissions of tyre and brake wear are combined because these fall in the same NFR[4] category for inventory reporting and cannot be separated out in the data reported by each country. The trends are very similar for the UK, Germany, France and Denmark with significant reductions in exhaust emissions and a growing share in the contribution of NEE from 2000 to 2016.

[3] Emissions reported by each country under the UNECE LRTAP Convention can be download from https://rod.eionet.europa.eu/obligations/357/deliveries

[4] NFR stands for Nomenclature for Reporting and is the UNECE source code system for reporting of air pollutant emissions. NFR 1A3bvi is the source code for tyre and brake wear emissions combined. Emissions from road abrasion are reported separately under NFR 1A3bvii.

A closer inspection of the emission inventories for each country shows that the overall PM_{10} mg/km emission factors for all road traffic NEE sources are broadly the same for the UK as for each other country, but are somewhat lower for the Netherlands. However, the bigger differences between the NEE inventories for the Netherlands compared with the UK and other countries is in the ratio of $PM_{2.5}/PM_{10}$ assumed for each NEE source which, as stated earlier, are much lower in the Netherlands inventory compared with the figures in Table 3 used by other countries. Given the exhaust emission factors used by the Netherlands are similar to values used in the UK and other countries, a combination of somewhat lower tyre wear and brake wear factors for PM_{10} combined with lower $PM_{2.5}/PM_{10}$ ratios explains the differences in the contribution of NEE sources to overall traffic emissions of PM_{10} and $PM_{2.5}$ implied by the Netherlands inventory compared with other countries including the UK, as shown in Figure 8. According to the report on the Dutch emissions inventory (Klein et al, 2018), the emission factors for tyre wear are based on the mass loss of tyres resulting from the wear process and are derived from Ten Broeke et al. (2008). It is assumed that 5% of the tyre particulate matter emissions can be considered to be PM_{10}, the rest are larger fragments that do not stay airborne but are deposited to the soil or surface water. For brake wear, the emission factors are derived from RWS (2008). It is assumed that the material emitted from brake linings is 49% particulate matter (PM_{10}) and 20% are larger fragments. The remainder of the material (31%) remains on the vehicle. Thus, whilst the overall trend in emissions from NEE sources is similar for PM_{10} in the Netherlands compared with other countries, they show a much weaker share in NEE for $PM_{2.5}$ because of the low $PM_{2.5}/PM_{10}$ assumed for these emissions.

The inventories for Sweden and Finland are different with a much greater contribution of road abrasion to the PM_{10} inventory as a consequence of accounting for the effect of studded tyres. However, in all countries the inventories are reporting an ever increasing proportion of NEE to overall transport emissions.

These inventories clearly illustrate a common theme that further reductions in PM emissions from road transport in all European countries will be limited if no further actions are taken to reduce the non-exhaust emissions.

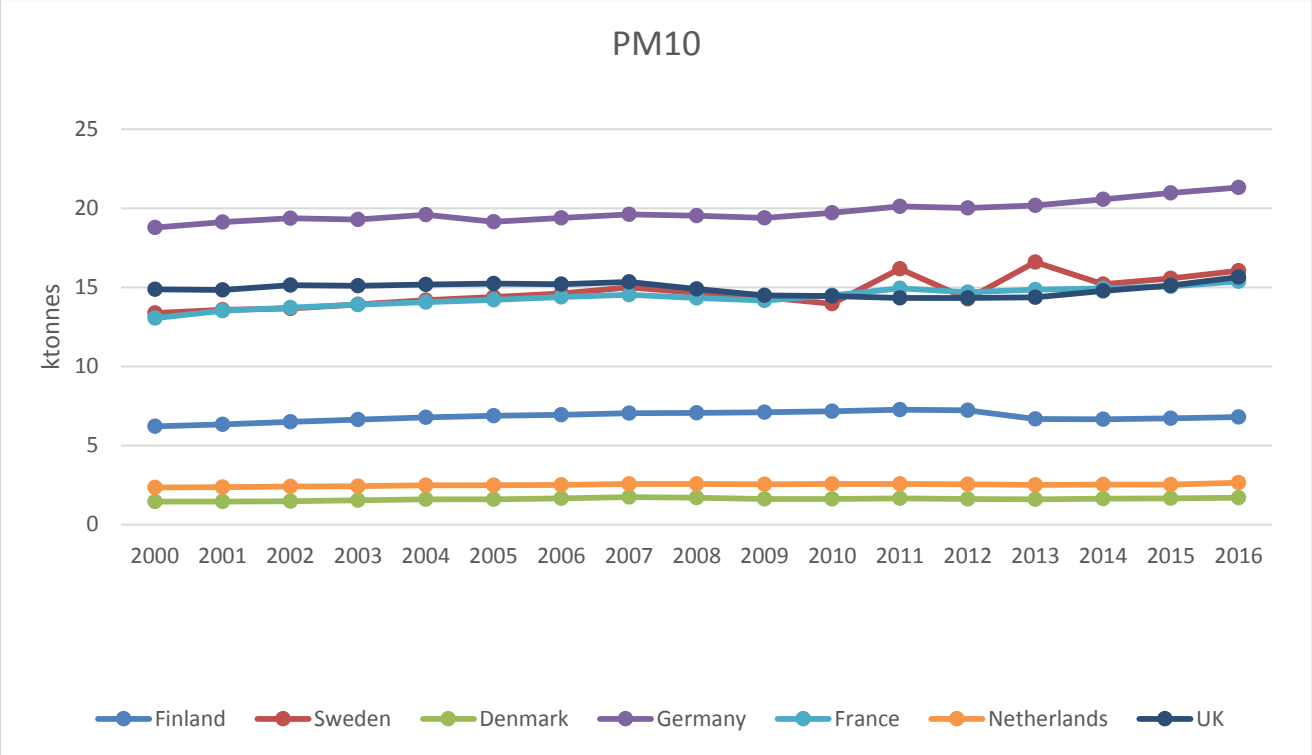

Figure 6: Non-exhaust emissions of PM_{10} from road transport tyre and brake wear, and road abrasion, according to the emission inventories submitted by countries to UNECE CLRTAP in 2018.

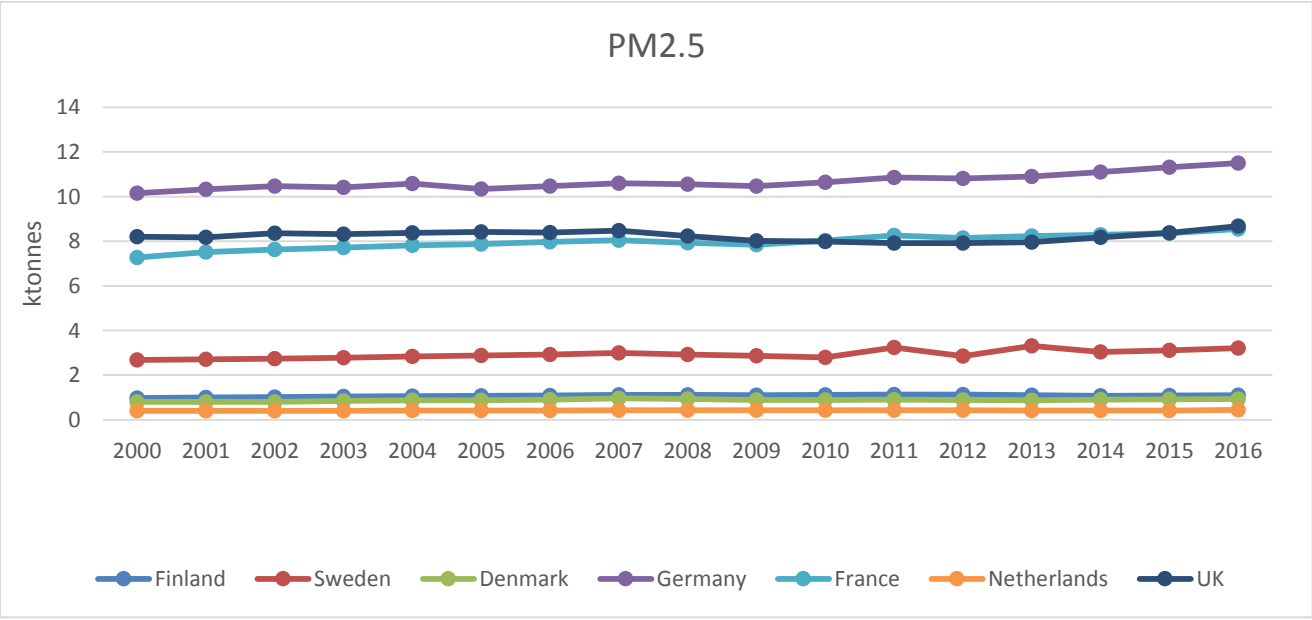

Figure 7: Non-exhaust emissions of $PM_{2.5}$ from road transport tyre and brake wear, and road abrasion, according to the emission inventories submitted by countries to UNECE CLRTAP in 2018.

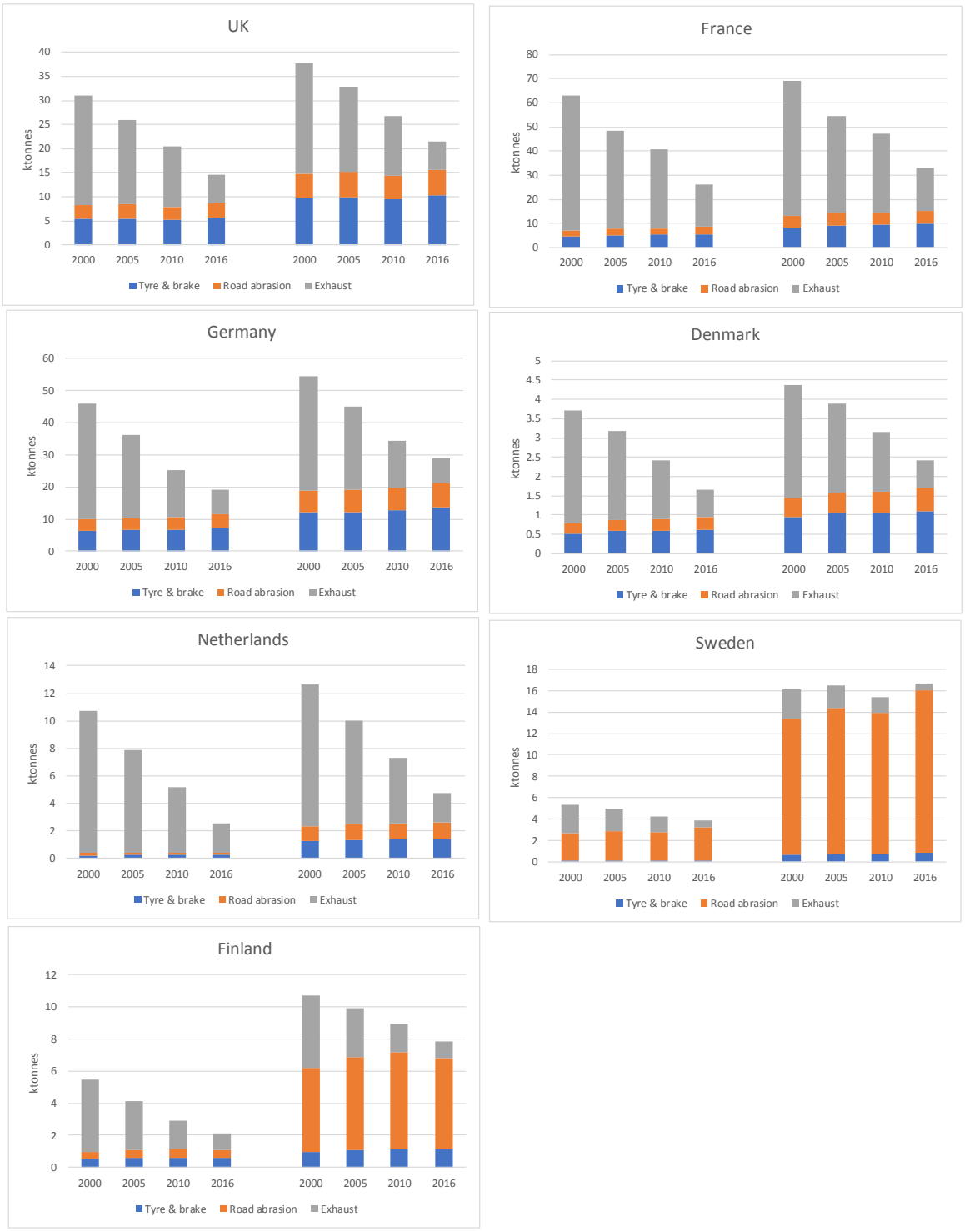

Figure 8: Non-exhaust and exhaust emissions of $PM_{2.5}$ (left hand set of bars) and PM_{10} (right hand sent of bars) from road transport according to the emission inventories submitted by countries to UNECE CLRTAP in 2018.

2.1.4 Non-exhaust emissions of metals and PAHs

Tyre and brake wear are sources of various metals and polyaromatic hydrocarbons (PAHs). The EMEP/EEA Emissions Inventory Guidebook provides average metal contents of tyre and brake wear material that are used with the PM inventory to estimate emissions for each metal. The metal contents in the Guidebook cover a wide range and are taken from various literature sources, many of which are quite old, the most recent source being for 2008.

The NAEI uses older sources of data on metal contents to estimate emissions of cadmium, chromium, copper, nickel and zinc from tyre and brake wear. Of these metals, only the reporting of cadmium is mandatory for reporting under CLRTAP, the other metals are reported voluntarily.

*Table **4*** shows the contribution of the tyre and brake wear source as a percentage of total UK emissions of each metal in 2016. There are no metal emission factors available for road abrasion. The contributions of these sources are likely to be higher in urban areas with large traffic volumes.

Table 4: Contribution of tyre and brake wear sources from road transport to total UK emissions of metals in 2016.

	Cd	Cr	Cu	Ni	Zn
% NEE	0.8%	3%	47%	0.8%	21%

The metal factors used by the NAEI are different to the ones in the Guidebook and it is estimated that if the Guidebook factors were used for tyre and brake wear, their contribution to total UK emissions would be higher than current figures in *Table **4*** suggest for Cd, Cr and Cu. In the case of Cu emissions, the contribution of non-exhaust sources would be as high as 90%. This would be consistent with the estimates made for the UK in the study by Denier van der Gon et al (2007), referred to in Section 4.3.

Most countries use the Guidebook factors for compiling inventories for these sources, but some countries use metal contents derived from country-specific information from industry or national research sources. For example, Scandinavian countries base their inventories on Swedish and Norwegian studies on tyre and brake wear rates and compositions (for example, see reference to Sweden's inventory report given in Section 2.1.1). For brake wear, account is taken of different metal contents of branded brake linings compared with linings from independent suppliers and how these have changed over time since the 1990s. Assumptions were used on the proportion of vehicles using branded vs independent brake linings

depending on the age of vehicle. The Zn content is said to have decreased since the 1990s on both types of linings. Denmark uses a mix of factors from the Guidebook and information from the Danish Tyre Trade Environmental Foundation (Winther and Slentø (2010).

The EMEP/EEA Emissions Inventory Guidebook provides the average content of three PAH species in tyre wear and brake wear which some countries use for their inventories. These are for benzo(a)pyrene, benzo(b)fluoranthene and benzo(k)fluoranthene, which are three of the four PAHs included in the UNECE Persistent Organic Pollutants (POPs) Protocol. The Guidebook acknowledges that these factors are from a very old study based on a single tyre type and brake pad. Most countries assume that the PAH factors remain constant in time such that the PAH emissions inventory generally rises over time since 1990 in line with the projections for growth in traffic. However, France and the Netherlands assume a large decrease in the PAH inventory for this source from 2010 due to implementation of the EU REACH Regulation which prohibits the use of so-called "PAH-rich" extender oils in tyres produced after January 2010[5].

The UK reports inventories for black carbon (BC) to the UNECE CLRTAP on a voluntary basis using simple BC/PM fractions given in the Guidebook. The factors are given as a fraction of $PM_{2.5}$ mass emissions and vary from 0.15 for tyre wear to 0.03 for brake wear and 0.01 for road abrasion, but all BC factors have high uncertainty ranges.

2.1.5 Spatial distribution of non-exhaust emissions of PM in the inventory

Since the inventory methodology for NEE is based on simple vehicle-specific and speed-dependent (in the case of tyre and brake wear) g/km emission factors, the NAEI distributes UK non-exhaust emissions of PM_{10} and $PM_{2.5}$ using a 1 km x 1 km resolution map of estimated total vehicle kilometres on major and minor roads. The most recent published maps of UK emissions are for the year 2015 (http://naei.beis.gov.uk/data/mapping). This approach will obviously lead to the largest emissions along the roads with the highest traffic flows and lowest average speeds.

Although there are currently no alternative methods for mapping NEE, this is a fairly crude approach because the same emission rates are stretched out along the entire length of road within an allocated speed band but it does not take into account specific traffic or road features,

[5] https://echa.europa.eu/documents/10162/13577/enforcement_ws2_ap_6_a_en.pdf/d64d1794-f70a-4cb8-b376-666966a30a0b

including gradients that might lead to locally high emission events. This is particularly relevant to emissions from brake wear where emissions would be expected to be highest at locations with high intensity braking events and near zero where traffic is flowing freely and no braking occurs. Emissions from tyre wear and road abrasion might be more evenly distributed although even here there is the likelihood of higher wear rates occurring along stretches of roads with significant acceleration and deceleration events.

Research is being undertaken to better understand the dynamics of braking and the effect this has on wear rates and PM emissions, as represented in presentations given at the Eurobrake conference at The Hague in 2018[6], although this has not yet led to a more refined approach for modelling the spatial variation in brake wear emissions. Work being carried out for the German auto industry is developing a high resolution brake use inventory for spatially resolving emissions of PM from brake wear. The initial phase of the development used high frequency dynamic vehicle measurements data to produce a proxy for brake intensity, i.e. negative Vehicle Specific Power which showed how highest intensity braking and therefore high brake wear emission rates is likely to be at specific places such as motorway exit roads (VDA, 2017). Work is currently underway to model the effect that such high intensity braking and emission rates might have on local roadside concentrations of PM from this source.

2.2 Modelled relative amounts of brake and tyre non-exhaust emissions across the UK major road network

Emissions modelling using published emission factors and traffic assumptions has been undertaken to understand the relative distribution of brake wear and tyre wear non-exhaust emissions across the UK road network. Emissions of non-exhaust PM were calculated for the UK major road network and for major roads in London. Speed dependencies for wear emissions were calculated as per the emissions guidebook (EMEP/EEA, 2009). The emissions for each non-exhaust source are discussed briefly. For each NEE source category, emission rates along each road link were banded into quartiles to provide a more helpful graphical summary. (Note that road links have variable lengths and are generally shorter on urban roads and longer on roads between urban centres).

[6] http://2018.eurobrake.net/programme/technical-programme

2.2.1 Brake wear PM

Modelled emissions of brake wear are shown in Figure 9. This clearly shows the dominance of brake wear emissions in urban areas, especially in London and within the M25 but also in Newcastle, Sunderland and Glasgow. Upper quartile emissions can also be seen on road links around smaller conurbations including Stoke-on-Trent, Plymouth, Cambridge, Southampton, Portsmouth and Bournemouth and in north and south Wales.

Looking at the enlarged section for the London region in Figure 10 it is clear that much of the capital's road network has modelled brake wear emissions in the upper two quartiles of road links of UK emissions. Roads with brake wear in the top quartile include much of the M25 and North Circular, major roads in west London and also some busier roads in the centre.

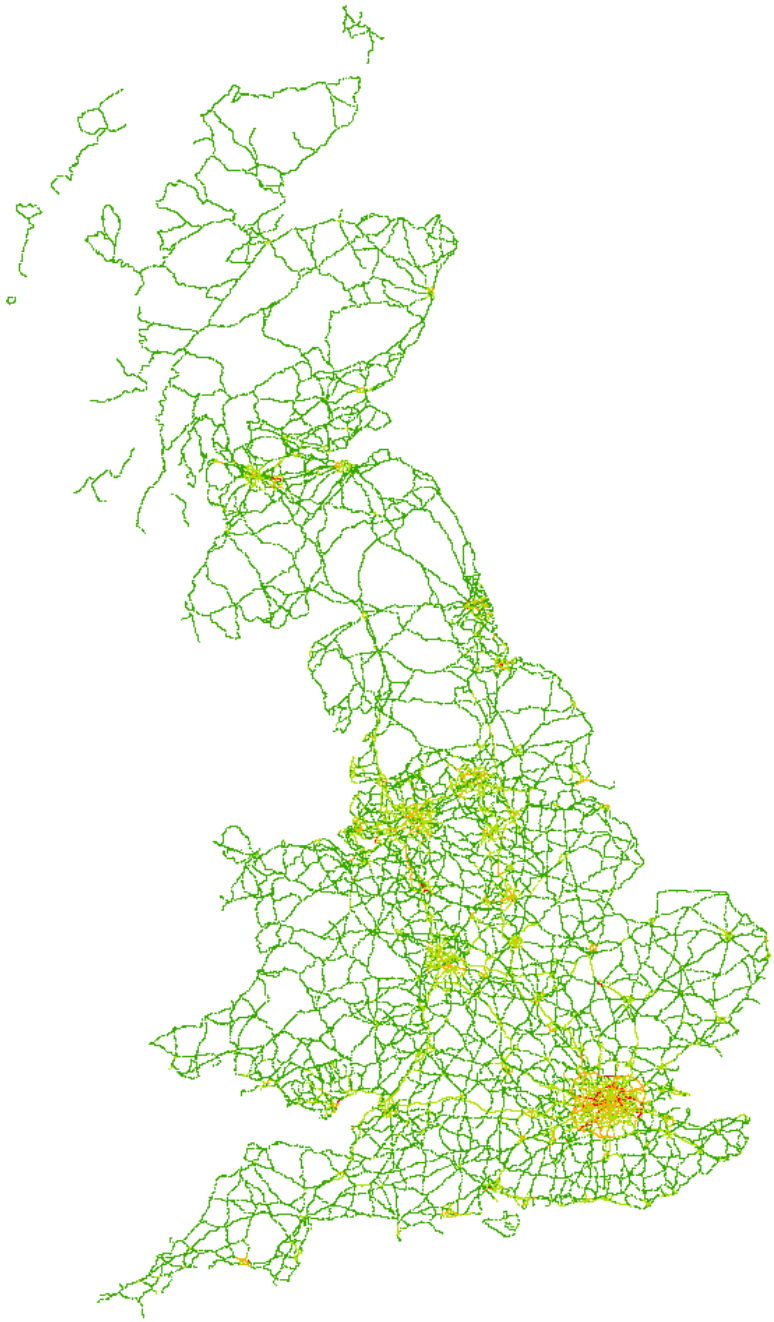

Figure 9: Emissions of PM_{10} brake wear for the UK major road network. The emissions from all the UK major road links have been divided into quartiles (from low to high these road links are coloured dark green, light green, orange and red).

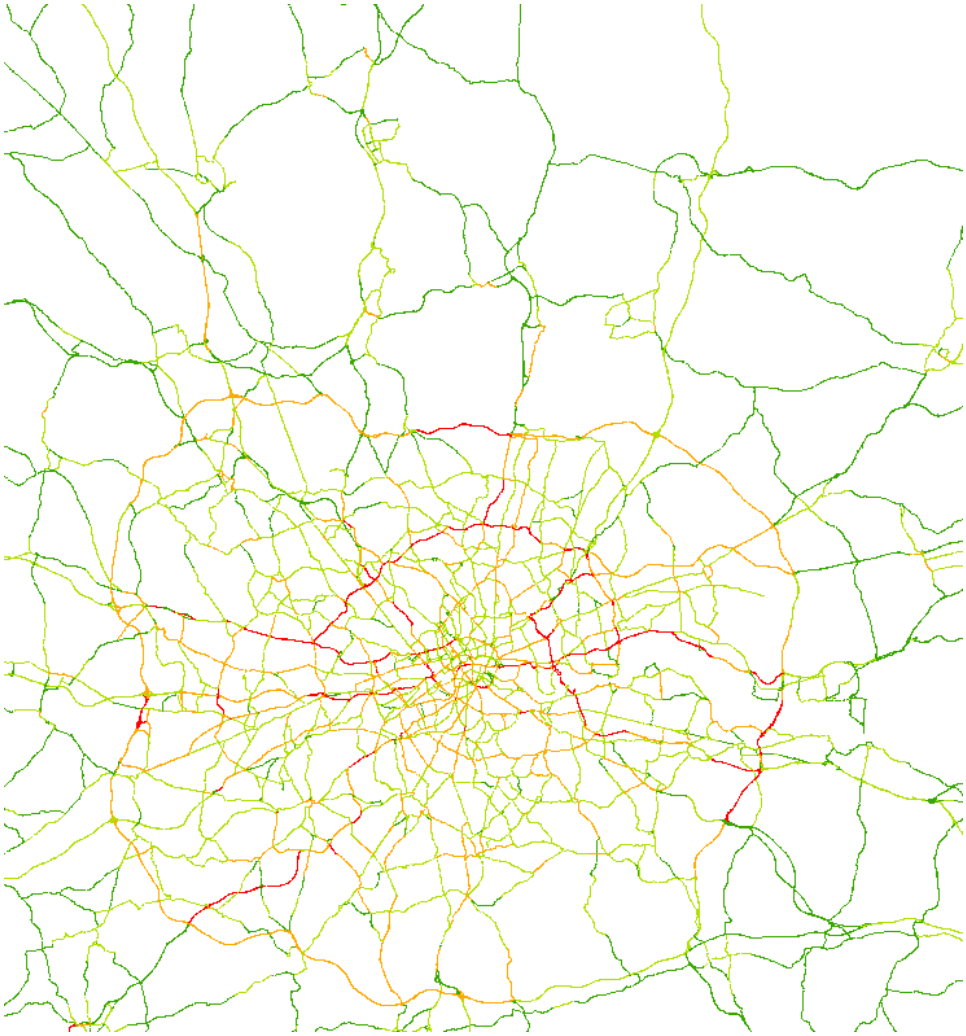

Figure 10: Enlarged section of Figure 9 showing emissions of PM_{10} brake wear for the major road network around London. The emissions from all the UK major road links have been divided into quartiles (from low to high these road links are coloured dark green, light green, orange and red).

2.2.2 Tyre wear PM

Modelled emissions of tyre wear are shown in Figure 11 and show the dominance of emissions from sections of the motorway network, most especially the M25 and the motorway networks emanating from Birmingham, Bristol, Leeds, Liverpool, Manchester and Southampton. Outside the motorway network top quartile emissions can also be seen along the A14 in Cambridgeshire.

Looking at the enlarged map of the London region shown in Figure 12, modelled emissions are clearly greatest along multiple carriageway roads such as the North Circular, M4, A40, A12 and A2 in addition to the M25 and feeder motorways.

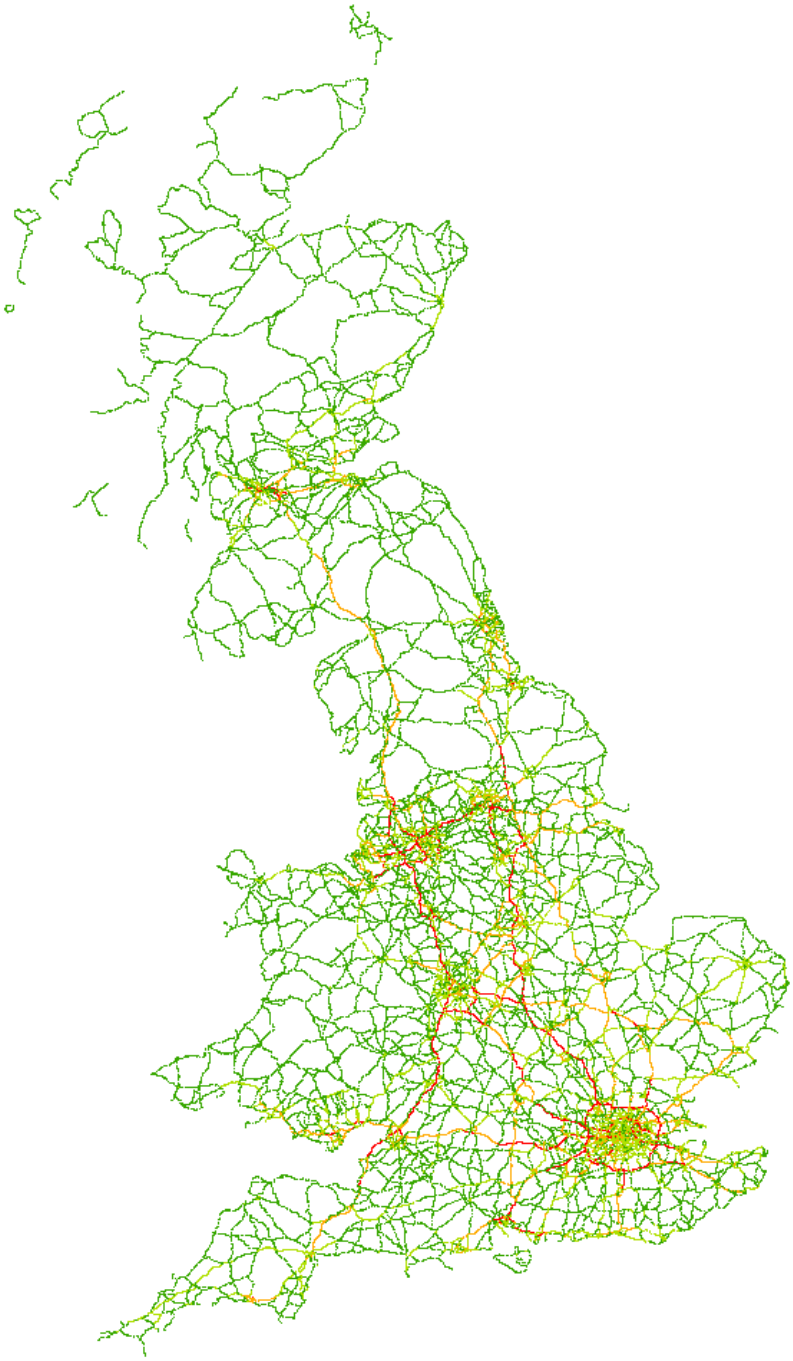

Figure 11: Emissions of PM_{10} tyre wear for UK major road network. The emissions from all the UK major road links have been divided into quartiles (from low to high these road links are coloured dark green, light green, orange and red).

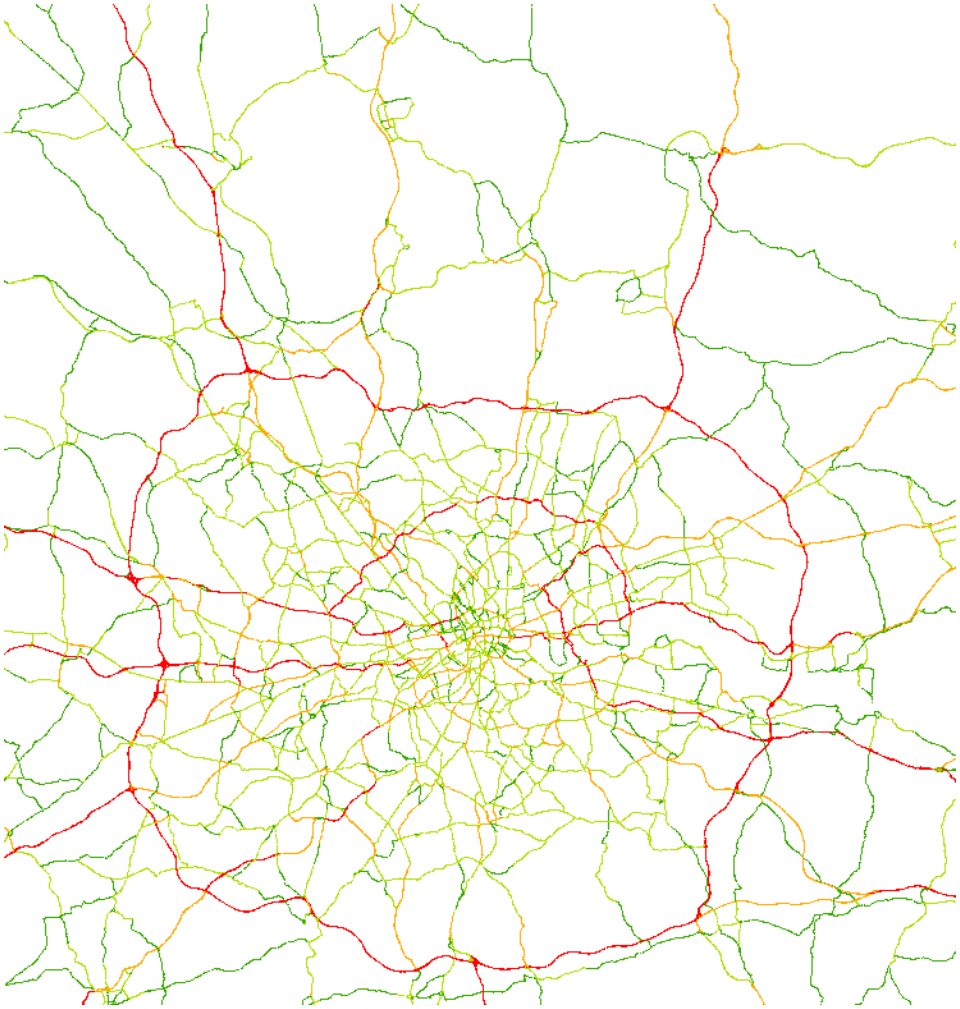

Figure 12: Enlarged section of Figure 9 showing emissions of PM_{10} tyre wear for the major road network around London. The emissions from all the UK major road links have been divided into quartiles (from low to high these road links are coloured dark green, light green, orange and red).

2.2.3 Discussion

Tyre wear emissions were greatest along the UK motorway network. This contrasts with emissions from brake wear that were dominated by that from major roads in urban areas. These variations can be understood in terms of the different factors that contribute to emissions calculations.

Both brake wear and tyre wear are dependent on vehicle speed in addition to having dependences on vehicle type and flow (EMEP/EEA 2009). Speed dependency also differs between the two wear sources. As shown in Figure 13 emissions of brake wear increase by a factor of around nine at urban driving speeds and conditions compared to emissions from fast

free-flowing-traffic motorways; this compares to a factor of 1.5 between these road conditions for tyre wear. This relative speed dependency leads to proportionally greater urban emissions of brake wear relative to that from tyres.

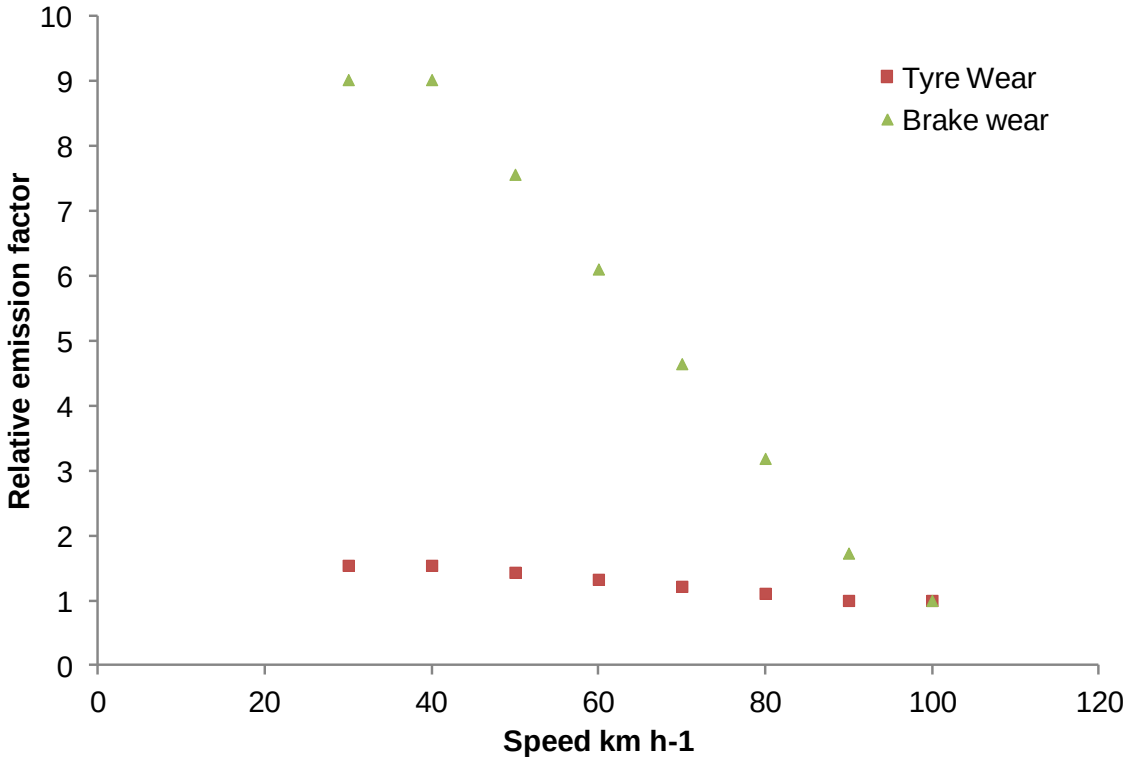

Figure 13: Speed dependence of emissions factors for brake and tyre wear. Each are normalised to their respective emissions rate at 100 km h^{-1}. In this plot speed refers to the average traffic speed and not the instantaneous speed of any vehicle.

These modelled emissions for non-exhaust PM_{10} show clear variation across the UK road network in the amounts contributed from the different sources of NEE. The best locations for measurements of NEE contributions may not necessarily be coincident with current air quality monitoring sites. Candidate locations for experimental campaigns to measure non-exhaust PM should therefore focus on the trunk and motorway road network. Upper quartile emissions from NEE sources can also be found on urban trunk roads. In many cases these locations will also be sources of important population exposure. This analysis has, however, not considered minor roads, which may also be important sources of population exposure to NEE where these have lots of start/stopping traffic and speed bumps, for example.

2.3 The UN Particle Measurement Programme Non-Exhaust Emissions IWG

The Particle Measurement Programme (PMP) is an informal working group (IWG) reporting to the UN Transport pollution and Energy group (GRPE) that includes representatives from the automotive industry, government, the Society of Automotive Engineers and the International Organization for Standardization. At the request of GRPE, the PMP has been looking into non-exhaust emissions. The following is edited information from the PMP Non-Exhaust Emissions IWG response to the joint Defra and DfT call in July 2018 for evidence on brake, tyre and road surface wear (seen by AQEG with permission of the PMP)[7].

PM_{10} emission factors of 2-13 mg per vehicle-km have been reported for passenger cars for both brakes and tyres (Grigoratos and Martini, 2015; Denier van der Gon et al., 2018) and 0-8 mg per vehicle-km for road abrasion (Gustafsson, 2018). Available emission factors are not comparable to each other and do not provide accurate information on emissions from different vehicle classes and different types of tyres and brakes.

Measured emission factors for brake-wear strongly depend amongst other factors on the testing method (pin-on-disc, brake dyno, chassis dyno, on-road test), the duty cycle used and the measurement setup. The PMP IWG on Non-Exhaust Emissions is working on developing a common method for measuring both particle mass and particle number brake wear emissions using a sophisticated duty cycle informed by real-world driving conditions that, crucially, will also permit collection of data for emissions from different types of brakes and brake materials, and possibly also from different vehicle classes. A measurement method via brake dyno is being developed.

For tyre wear particle emissions, no common, robust, reliable and repeatable method exists to determine emission factors. The European Commission recently mandated the development of an experimental method for the measurement of tyre abrasion rate (mg/km) as part of the Tyre Labelling Regulation (COM(2018) 296). Methodology should also include determination of the emissions of the PM_{10} and $PM_{2.5}$ (and particle number) fractions specifically per tyre-km. It is not currently clear who will take on this activity and it will take time for a method to be developed. The relationship between tyre wear and type of road surface needs to be investigated together.

[7] https://consult.defra.gov.uk/airquality/brake-tyre-and-road-surface-wear/

The PMP IWG's view from industry data not yet publicly available is that reductions in brake wear particle mass emissions from vehicles substantially using regenerative braking will be much larger than any increases in brake wear emissions associated with increased vehicle mass. Such vehicles are also expected to have lower brake wear particle number emissions since less use of friction brakes results in an overall cooler brake system which lowers particle formation. On the other hand, there is strong indication that heavier vehicles will emit greater tyre wear particle mass and particle number, e.g. Foitzik et al. (2018).

3 The Measurement Evidence for Non-Exhaust Emissions

Non-exhaust emissions of PM have been characterised in the laboratory and also by sampling air from near the tyres / brakes using equipment mounted on individual test vehicles. Most ambient measurement evidence of the importance of non-exhaust emissions from vehicles stems either from size-segregated observations at the roadside concentration compared with the urban background, or from source apportionment based on chemical composition.

Ambient levels of particles originating from NEE are controlled not only by the source strength, but also by dispersion, i.e. the efficiency for the emitted particles to be diluted through turbulence. Emissions of particles from the abrasion of tyres, brakes and the road surface contribute directly, but they also deposit back onto the road surface, mainly due to gravitational settling, from which they can then be re-suspended, either through the turbulence of the passing vehicles (vehicle-induced resuspension) or by wind (wind-driven resuspension). However, particles from non-traffic sources deposited or otherwise transported onto road surfaces also contribute to road dust emissions.

Padoan and Amato (2018) have recently reviewed the global literature on the contribution of vehicular NEE sources to atmospheric concentrations; amongst these studies there are only few measurement derived estimates for the UK.

3.1 Evidence for NEE from size-resolved PM measurements

Friction processes generally generate particles with a diameter that exceeds 1 μm and this also applies to the brake and tyre wear (Grigoratos and Martini, 2015). Resuspension is also efficient only for supermicron particles. By contrast, combustion processes such as those responsible for the exhaust emissions produce sub-micron particles. Thus, by distinguishing between the aerosol mass <1 μm (PM_1) and the mass contained in the size range between 1 and 2.5 μm (or between 1 and 10 μm) the contribution from exhaust and non-exhaust vehicle emissions to ambient concentrations of $PM_{2.5}$ (PM_{10}) can be quantified, with the caveat that it will contain some re-suspended road dust not originally associated with vehicle emissions. Overall, it has been found that the NEE component is most commonly associated with coarse particles (i.e. PM_{10} rather than $PM_{2.5}$) in close proximity to the sources (Lough et al., 2005). Unfortunately, PM_1 is rarely measured and the more commonly monitored $PM_{2.5}$ and PM_{10} are used to calculate NEE represented by the increment of the coarse component ($PM_{coarse} = PM_{10} - PM_{2.5}$) at the roadside compared to the urban background. It should be noted though that a

component of the roadside $PM_{2.5}$ also includes some NEE in the size range 1 to 2.5 μm, which will be unaccounted for.

In a European meta-analysis of $PM_{2.5}$ and PM_{10} concentrations, Hopke et al. (2018) showed that whilst PM_{coarse} accounts for 34% of the PM_{10} at suburban and urban background sites, it accounts for 43% of the PM_{10} at traffic sites across sites, with the values for the UK (and Scandinavia) lying at the top end of the range (50% to 66%), which the authors attribute to winter road sanding that is more prevalent in N Europe. More generally, comparing multiple datasets, it is evident that emissions vary according to conditions and habits in different countries, such as drier roads in the Mediterranean or the used of studded winter tyres in Scandinavia causing more road wear and dust (Amato et al., 2014).

By combining size-distribution measurements with chemical tracer information, Harrison et al. (2012) estimated that at London Marylebone Road the roadside increment in super-micron PM is composed of 55% of brake dust, 38% resuspended dust and 11% tyre dust. Lenschow et al. (2001) found that 55% of the PM_{10} roadside increment on a busy street in Berlin represented exhaust emissions and tyre abrasion and the remaining 45% resuspended road dust. Similarly, Querol et al. (2004), analysing PM_{10} and composition data from a range of European cities found that, typically, exhaust and non-exhaust emissions such as resuspension and abrasion each contribute about half of the roadside increment.

Correlations between the roadside increments in PM_{coarse} and gaseous tracers have been observed during several studies. For example, Harrison et al. (2001) reported a linear increase between the increment of Marylebone Road PM_{coarse} and the increment in NOx, where London Bloomsbury was selected as the urban background site against which the increments were referenced.

3.2 Evidence for NEE from PM chemical composition

Without chemical information, it is difficult to distinguish between the different types of NEE emission. A number of studies have reported ambient concentrations of NEE particles on a composition basis. Because a large portion of the mass emitted is refractory and not water soluble, many common methods of ambient PM composition monitoring such as ion chromatography, gas chromatography and Aerodyne Aerosol Mass Spectrometry are of limited use. Instead, most atmospheric composition studies have targeted the metallic component of NEE particles, which is also an aspect of concern regarding toxicity. Common

methods of analysing filter or impactor samples offline include various forms of x-ray spectroscopy (e.g. x-ray fluorescence, particle-induced x-ray emission) or analysis with inductively coupled plasma mass spectrometry/optical emission spectroscopy (ICP-MS/OES) after acid digestion. The Defra Heavy Metals Network routinely monitors a wide range of metals in ambient air at 24 sites by this method.

In addition to offline analysis, online methods also exist. The XACT is a relatively new instrument capable of semi-continuous x-ray florescence analysis for metals (Furger et al., 2017), which allows for a continuous, high time resolution (~1 hour) dataset to be generated. Laser-ablation single-particle mass spectrometers such as the Aerosol Time of Flight Mass Spectrometer (ATOFMS) are capable of identifying NEE particles on an individual level, based on the presence of key ions (Beddows et al., 2016; Dall'Osto et al., 2014). Because these are real-time instruments, very high time resolution is possible; however, because the composition data are not strictly quantitative, single-particle mass spectrometers are more suited to delivering number concentrations segregated by particle type rather than quantifying PM according to mass.

Various elements have been detected in the atmosphere that are known to be present in NEE, in particular Cu, Sb and Ba from brake wear, Zn for tyre wear, crustal elements such as Al, Ca and Si associated with dust, and some elements such as Fe that can be associated with more than one (Thorpe and Harrison, 2008; Pant and Harrison, 2013).

However, positively associating an observation of these with NEE is not always straightforward because these elements can also be present in other sources, such as non-road, wind-blown dust and industrial sources. Also, the composition of braking components varies across the vehicle fleet and detailed information of component composition is often proprietary and not known.

However, as with the particle size measurements described above, source apportionment can be achieved by comparing a roadside or urban site against a background site (Geitl et al. 2010; Harrison et al. 2012), comparing the composition of particulates at the exit of a road tunnel with those at the entrance (Sternbeck et al., 2002; Lough et al., 2005) or inspecting the ratios of elements and comparing to those expected based on emissions studies (Weckwerth, 2001).

In the study by Gietl et al. (2010), a range of trace metals in the PM_{10} size fraction were measured at roadside on Marylebone Road and at a nearby background site in Regents Park. Taking the difference between Marylebone Road and Regent's Park to indicate the roadside

increment in concentration, an elevation was seen in the concentrations of the following metals: Al, Sb, Ba, Ca, Cu, Fe, Mg, Ti, V and Zn. The enrichment in Al, Ca and Mg is probably largely attributable to resuspended road dust, and tyre wear particles are generally considered to be the main source of Zn in the roadside environment. The other elements, and especially Ba, Sb and Cu, are typically attributable to brake wear, with this being the predominant source of Cu in the roadside environment (Section 2.1.4). Harrison et al. (2012) similarly observed strong correlations between Fe and Cu, Sb and Ba, strongly suggestive of a common source.

Further measurements of trace metals at roadside on Marylebone Road (data supplied by Dr Paul Quincey, National Physical Laboratory) showed only small changes in concentration between the annual mean in 2011 and that in 2017. Temporal trends over this period in Cd, Co, Se, As, Cr, Pb, Mn and Fe were broadly flat while Cu, Ni and V showed an apparent downward trend, with a number of elements including Cu, Zn, Mn and Fe showing a small increase between 2016 and 2017.

Source attribution based on metal concentrations, generally speaking, suggests that brake wear and resuspension are more significant than tyre wear. Based on the data of Harrison et al. (2012), Padoan and Amato (2018) derived aerosol contributions (in the size range 0.9 to 11.5 μm) of 3.3, 2.3 and 0.65 μg m^{-3} for brake wear, road dust and tyre wear, respectively. However, it must be noted that these data relate to urban roadside and not to motorways which, as shown in section 2.2.2, are where the tyre wear NEE are expected to be greatest.

3.3 Measurement-based source receptor modelling

Receptor modelling refers to methods used to infer the contributions of different sources to measured concentrations of air pollutants. It has been widely applied to airborne particles and depends upon the fact that particles arising from different sources are of differing chemical composition.

In its simplest form, receptor modelling uses a single chemical tracer to relate particles to a specific source. Use of this method depends upon an assumption that the chemical tracer is specific to the one source of particles and that the ratio of the mass of the total particle to that of the chemical tracer is known. An example is the use by Gietl et al. (2010) of Ba as a tracer for brake wear particles in roadside air. The choice of Ba was made on the basis of its known presence in brake pads, and measured concentrations made at roadside and in the urban background, which showed a strong traffic increment at roadside. Ba concentrations at

roadside when corrected for the background contribution correlated strongly with a number of trace elements including Fe (which derives from wear of both brake pads and discs), indicating that they had a common source. There was no indication of an appreciable contribution to Ba other than from the brake wear emissions. In order to make use of Ba as a quantitative tracer, it was necessary to estimate its abundance in brake wear emissions, and that was accomplished by use of emission factors for brake wear particle mass combined with traffic volumes to estimate the total emissions per kilometre of brake wear particles on Marylebone Road during the measurement period. An alternative might have been to collect brake wear particles in the laboratory and determine their Ba content, but the literature has shown that the elemental content of brake pads is immensely variable (Hulskotte et al., 2014), and consequently collection of a small number of samples from different brake pads would not be representative. Inferring the elemental abundance from atmospheric data ensures that a fleet average value is obtained. In a slight variation on the Ba tracer method, Harrison et al. (2012) used a combination of elemental and particle size distribution data to infer the contributions of brake wear, tyre wear and resuspension to airborne particulate matter on Marylebone Road using Ba, Zn and Si respectively as tracers for brake wear, tyre wear and resuspended road dust.

In general terms, tracer-based receptor modelling for quantitative source apportionment of PM is challenging for NEE because the composition profiles of the components are highly variable. Resuspended road dust is particularly problematic because in addition to the composition of natural dust varying with local mineralogy, the dust is often heavily contaminated with brake and tyre wear (Adamiec et al., 2016), meaning that the definitions of the different NEE become blurred. Emissions can also vary according to a number of other factors such as brake pad formulation, driving conditions and even humidity (which affects resuspension). This means that quantification based on these methods can be highly uncertain (Pant and Harrison, 2013).

As an alternative approach, Positive Matrix Factorization (PMF) can also be used, which is an algorithm that requires no *a priori* assumptions regarding composition. Instead, this identifies 'factors' in the data corresponding to collective variations in multivariate datasets and has been successfully applied to this problem in certain environments (Fabretti et al., 2009). Generally, this method performs best when using high time resolution data (from a semi-continuous analyser or automated sampler), as variations with time of day brought on by changes in traffic activity ensure for better differentiation from other sources. Most commonly it is applied to elemental and ionic concentrations in airborne particles and identifies discrete factors whose chemical profiles can be used to relate them to specific sources of particles.

PMF performance varies according to specific environments and data sources and it cannot be reliably predicted in advance what factors a PMF will identify for a given dataset, but many recent well-conducted studies have indeed been able to identify particles arising from vehicular non-exhaust sources, but are rarely able to disaggregate the non-exhaust particles according to their specific origins in tyre wear, brake wear or resuspension. This is because these tend to be temporally covariant and often, the method is not able to differentiate resuspended road dust from wind-blown soils which often have a similar chemical profile. Crilley et al. (2017) used high resolution metals data from the 2012 ClearfLo campaign along with data concerning particle size and mass and was able to identify NEE sources using PMF and, furthermore, estimated that at Marylebone Road, vehicle wear and resuspension contributed 13.4% (1.3 $\mu g\ m^{-3}$) and 31% (3.1 $\mu g\ m^{-3}$) of PM_{coarse}, respectively. These factors were differentiated by the greater content of Si, Al and Ti in the resuspended road dust as compared to high contributions to trace metal concentrations (Cr, Mn, Fe, Cu and Zn) from the vehicle wear particles (Ba was not measured). The Crilley et al. (2017) study identified factors associated with traffic emissions (exhaust and non-exhaust) and airborne soil in fine fraction particles at North Kensington and a traffic emissions factor in coarse particles collected at the North Kensington site.

Another factorisation using the Multilinear Engine (ME-2, a variant on PMF) was employed by Visser et al. (2015) to size-resolved elemental data, also obtained in London during ClearfLo, and was able to isolate three categories of NEE, each most prevalent within the PM_{coarse} size fraction and with a very strong roadside increment (through comparison with multiple sites) (Figure 14). These data are not full mass budgets, but they are illustrative of relative source strengths for different site types. Both the Crilley et al. (2017) and Visser et al. (2015) studies had ambiguities in the data, such as factors that could only be categorised as ‘traffic related’ or an inability to differentiate sea salt and road salt.

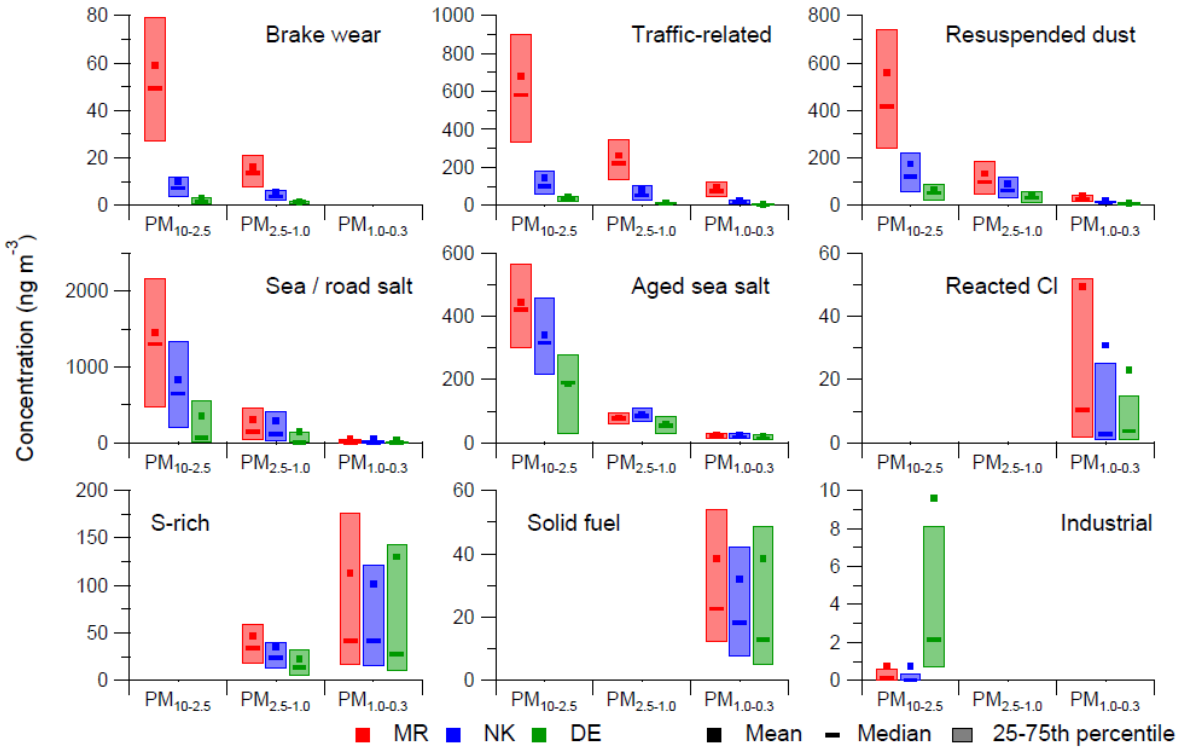

Figure 14: Comparison of different sources according to PM size fraction and site in the London area according to Visser et al. (2015). MR = Marylebone Road (roadside), NK = North Kensington (urban background), DE = Detling (rural background). Note that the mass concentrations only refer to the contributions to the measured elements, not a complete mass budget.

In an analysis of a different dataset also collected at North Kensington, Beddows et al. (2015) identified a factor attributed to non-exhaust and traffic/crustal particles in PM_{10} at North Kensington. This was identified from its characteristically high contribution to Al, Ca and Ti as well as to a number of trace elements associated with abrasion emissions. Beddows et al. (2015) also identified a factor attributed to traffic which was notable for its high contribution to Cu, Mo, Ba and Sb and a high concentration of Fe, both of which are features strongly associated with brake wear particles. Both factors made a significant contribution to concentrations of Zn which is normally taken as a tracer for tyre wear emissions in the absence of local industrial sources.

In a harmonised study of five major southern European cities, Amato et al. (2016) identified by PMF a factor showing broadly similar composition within all of the cities which they attributed to vehicular non-exhaust emissions which was additional to a vehicle exhaust factor. It showed chemical features typical of brake wear (Cu, Ba and Sb), tyre wear (Zn) and road dust (Si, Al and Ca). When the PMF outputs were quantified, the vehicle non-exhaust factor

accounted for between 8 and 14% of PM_{10} mass and between 1 and 8% of $PM_{2.5}$ mass. Averaged across the cities, the contributions of vehicle exhaust and vehicle non-exhaust particles to PM_{10} were very similar, while for $PM_{2.5}$, vehicle exhaust was dominant in four out of five cities.

The fact that the different non-exhaust particles sources have different size distributions was exploited in a study on Marylebone Road by Harrison et al. (2011). The Positive Matrix Factorization (PMF) method was applied to a dataset of hourly measurements of particle number size distributions collected at roadside on Marylebone Road. The data covered the range 15 nm to 10 µm in around 100 size bins. The data were analysed in conjunction with measurements of gas-phase pollutants and meteorological variables, and PMF yielded size distributions and associations with gaseous pollutants and meteorological data as well as diurnal patterns in particles of given size distributions. From these data, it was possible to identify two kinds of exhaust particles (nucleation mode and solid mode), brake dust and resuspension particles as well as another six factors associated with the urban background particulate matter.

3.4 Consistency between measurements and inventories for NEE

As a simple check on the relative magnitude of exhaust and non-exhaust emissions, data from London Marylebone Road and London North Kensington were analysed for the period 2013-2017. The analysis depends upon calculating the difference in annual mean concentrations between the two sampling sites which can be taken as approximately equal to the traffic increment arising from the traffic on Marylebone Road itself after subtracting the local background as represented by the concentrations at North Kensington. This is done for daily gravimetric measurements of $PM_{2.5}$ and PM_{10} and for elemental carbon (EC) and organic carbon (OC) measured daily by a thermo-optical technique. Conventionally, the sum of EC + 1.2 OC is taken as representative of the mass of carbonaceous particles from road traffic.

The annual mean concentration data for the two sites appear in Table 5, and in Table 6 the magnitude of the inter-site differences represented as Δ(LMR – LNK) and Δ(EC + 1.2 OC) are calculated. The analysis shows that the traffic increment of PM_{10} on Marylebone Road averages 10.3 µg m^{-3} over the five year period while the carbonaceous aerosol increment Δ(EC + 1.2 OC) amounts to 5.67 µg m^{-3}, around 55% of the PM_{10} increment. This therefore suggests that over the relevant period, non-exhaust emissions amounted to around 45% of total emissions of PM_{10} from road traffic on Marylebone Road. The data for Δ(EC + 1.2 OC)

appear to show a modest decline from 2013 to 2017, although this has not been tested for significance. Given that conditions on Marylebone Road may not be reflective of the entire UK, these data for PM_{10} are not inconsistent with the estimates of the relative magnitude of non-exhaust and exhaust emissions although it should be remembered that the PM_{10} increment at Marylebone Road contains road dust resuspension which is not included in the NEE estimates of the inventory.

It would be expected that the carbonaceous particles represented by Δ(EC + 1.2 OC) would be almost exclusively in the fine ($PM_{2.5}$) size fraction. It is therefore surprising that the average traffic increment of $PM_{2.5}$ over the five year period is 5.4 µg m^{-3} which is slightly lower than the traffic increment in the carbonaceous particles. This may be explained by the fact that traffic exhaust is not the only contributor to EC and OC which are present also in brake dust and tyre particles, as well as road dust. Nonetheless, these data suggest that the predominance of the non-exhaust particles is most probably within the coarse size fraction ($PM_{2.5-10}$) and that the overall magnitude of the NEE contribution to airborne particulate matter is broadly in line with the NAEI predictions when looked at relative to the vehicle exhaust contribution.

Table 5: Annual mean concentration (all in µg m^{-3}).

	MARYLEBONE ROAD				NORTH KENSINGTON			
	EC	OC	$PM_{2.5}$	PM_{10}	EC	OC	$PM_{2.5}$	PM_{10}
2013	4.52	6.21	17	29	0.85	3.74	12	19
2014	4.78	5.62	16	28	0.89	3.44	10	18
2015	3.94	5.57	14	26	0.75	3.12	9	16
2016	3.68	4.96	14	25	0.98	3.10	8	14
2017	3.39	5.34	14	25	0.87	3.16	9	N.D.

Table 6: Δ(LMR – LNK) (µg m^{-3}).

	EC	OC	$PM_{2.5}$	PM_{10}	Δ(EC + 1.2 OC)
2013	3.67	2.47	5	10	6.63
2014	3.89	2.18	6	10	6.51
2015	2.19	2.45	5	10	5.13
2016	2.70	1.86	6	11	4.93
2017	2.52	2.18	5	n.d.	5.14
Mean (2013-2017)	2.99	2.23	5.4	10.3	5.67

3.5 NEE contribution to urban-scale PM fluxes

It should be noted that these quantifications relate to the contribution of NEE to PM concentrations at ground level, a metric relevant for local human exposure. The atmospheric residence time of coarse particles is relatively short and many coarse particles will, e.g., re-circulate within street canyons and not be transported aloft. Thus the controls of NEE emissions to the overall emission from a city, for example, is very different, and this emission is what contributes to the national background and transports PM to vegetation. Making direct measurements of the fluxes of supermicron particle number above Stockholm, Vogt et al. (2011) found a relatively linear relationship between supermicron particle flux and CO_2 (as a tracer of fuel use) at moderate wind speed, whilst above 8 m s^{-1} super-micron particle fluxes increased strongly. This is consistent with similar measurements above Edinburgh, where supermicron particle fluxes showed a relationship with traffic activity only up to a wind speed of 7 m s^{-1}, above which they increased strongly with wind speed, with no identifiable (additional) relationship with traffic activity (Nemitz et al., 2000). The Stockholm measurements additionally demonstrated that the coarse particle fluxes were largest in spring and during that season they were also larger for dry than for wet roads (Vogt et al., 2011). This was attributed to emissions of road dust from studded tyre wear during the winter, following snowmelt.

However, whilst NEE net emissions may be largest during windy conditions, dispersion is also efficient during these periods. As a result, the contribution of NEE to local concentrations (and exposure) is usually limited. For example, Harrison et al. (2001) observed that whilst the contribution of $PM_{coarse}/PM_{2.5}$ ratio increased with wind speed, the absolute concentration of PM_{coarse} decreased.

4 Representation of Non-Exhaust Emissions in Models

4.1 Introduction

As NEE emissions factors have become available and the relative contribution of NEE to PM concentrations has increased, the inclusion of NEE in transport and dispersion models has become more routine. However detailed dispersion modelling assessments of the relative contribution of the different components of NEE near roads and their dependence on different vehicle fleets and/or road environments have not been conducted. (The use of source-receptor models to infer the contributions of NEE sources to measured concentrations of air pollutants have been discussed in Section 3.)

4.2 Dispersion Models

In dispersion models the source of road traffic emissions is generally assumed to be a line source with finite width and height to account for the uncertainty in the location of the vehicle exhaust system and for the initial dispersion immediately behind the vehicle. Sources of NEE and exhaust emissions of PM are treated in exactly the same way being included within the same road or line source. In some models, for example OSPM (Hertel et al. 1990) and ADMS-Urban (Hood et al, 2014), allowance is made for the effect of vehicle-induced turbulence on source parameters dependent on number, speed and cross-sectional area of vehicles but again exhaust and non-exhaust emissions are treated in the same way. With regard to NEE emissions tyre, brake and road wear are typically included, and sometimes also resuspension, depending on available emissions data. For example EMEP emission factors for tyre, brake and road wear have been applied (Barlow et al., 2007), and resuspension factors from the EMEP/EEA guidebook (EMEP/EEA, 2013). More recently the EMEP tyre and brake wear emissions factors have been adjusted to be in line with measurements recorded at London Marylebone Road (Harrison et al. 2012) and applied to the London Atmospheric Emission Inventory (LAEI, 2010). In the PCM model (Brookes et al. 2017), brake, tyre and road wear emissions have been taken directly from the NAEI and included in the dispersion modelling for the roadside increment and background models.

As an illustration of a calculation of impacts of NEE on PM concentrations, Figure 15 presents an example using the ADMS-Urban dispersion model. The model was used to calculate the contribution of the different components of NEE and exhaust emissions at five different

receptor locations (*Table **7***), for 2016 and 2020 (Hood et al., 2018). The emissions of NEE are based on the LAEI inventory with adjustments using Harrison et al. (2012) for tyre and brake wear. Total emissions of the components of NEE and also exhaust emissions of PM_{10} and $PM_{2.5}$ are shown in Table 8. It is seen that the adjustment has a large impact on brake wear emissions and that resultant total increases in total road traffic emissions of PM_{10} and $PM_{2.5}$ in 2016 are 65% and 53% respectively. For 2016, the 'background' concentrations have been calculated using monitoring data either from Rochester Stoke or Chilbolton depending on the wind direction. For 2020, background concentrations were derived from the 2016 data by multiplying by the ratio of the LAQM background map concentrations for the years 2020 and 2016 at the location of the rural background monitors. Background values for PM_{10} and $PM_{2.5}$ are 14.9 and 10.0 µg m^{-3} in 2016 and 14.4 and 9.6 µg m^{-3} in 2020.

Figure 15 shows NEE and exhaust contributions to PM_{10} and $PM_{2.5}$ at the five receptor locations. The greatest contribution at all sites was from the background in both 2016 and 2020, however the traffic contributions were important at all the roadside locations considered. Of the traffic contribution, NEE is greater than exhaust emissions in both 2016 and 2020 and its relative contribution was even greater in 2020 as exhaust emissions continue to decline. NEE shows at least as much variability between the sites as exhaust emissions. This is further illustrated in Figure 16 which shows the modelled spatial variation in $PM_{2.5}$ for 2016 with the congestion charging zone.

As is the case with most dispersion modelling this study does not include temporal variations in NEE emissions other than the typical diurnal pattern, for example the impact of rainfall events on the resuspension component of emissions (e.g. de la Paz et al. 2015). The modelling summarised in Figure 15 assumes that NEE follows the same diurnal profile as exhaust emissions with no seasonality, which may result in some bias in the reported apportionment. Dispersion modelling studies often seek to predict the contribution of road traffic to short-term events, for example during days when total PM_{10} concentrations exceed 50 µg m^{-3}. The source contributions due to resuspension during these days are a key feature of interest as evidenced by the examples reported in Section 5.3.

Table 7: Modelled receptors and details of nearest road flows and speed.

Receptor name	Annual average daily traffic (AADT)	% Light	% Heavy	Speed (km/h)
GN4 Greenwich - Fiveways Sidcup Rd A20	37,569	96%	4%	29
GR9 Greenwich - Westhorne Avenue	42,698	92%	8%	25
KC2 Kensington and Chelsea - Cromwell Rd	18,322	95%	5%	35
SK5 Southwark - A2 Old Kent Road	27,185	89%	11%	30
MY1/MY7 (FDMS) Westminster - Marylebone Road	58,718	90%	10%	20

Table 8: *Total road traffic emissions (t/yr) in the LAEI for 2016 and 2020 adjusted following Harrison et al. (2012). The figures in brackets are the percentage increase due to the adjustment.*

	Brake wear	Tyre wear	Road wear	Resuspension	Exhaust	Total
PM10 2016	1,967 (263%)	476 (9%)	426	451	395	3,714 (65%)
PM10 2020	2,023 (263%)	491 (9%)	439	455	208	3,616 (71%)
PM2.5 2016	787 (263%)	333 (9%)	230	0	375	1,724 (53%)
PM2.5 2020	809 (263%)	344 (9%)	237	0	197	1,587 (63%)

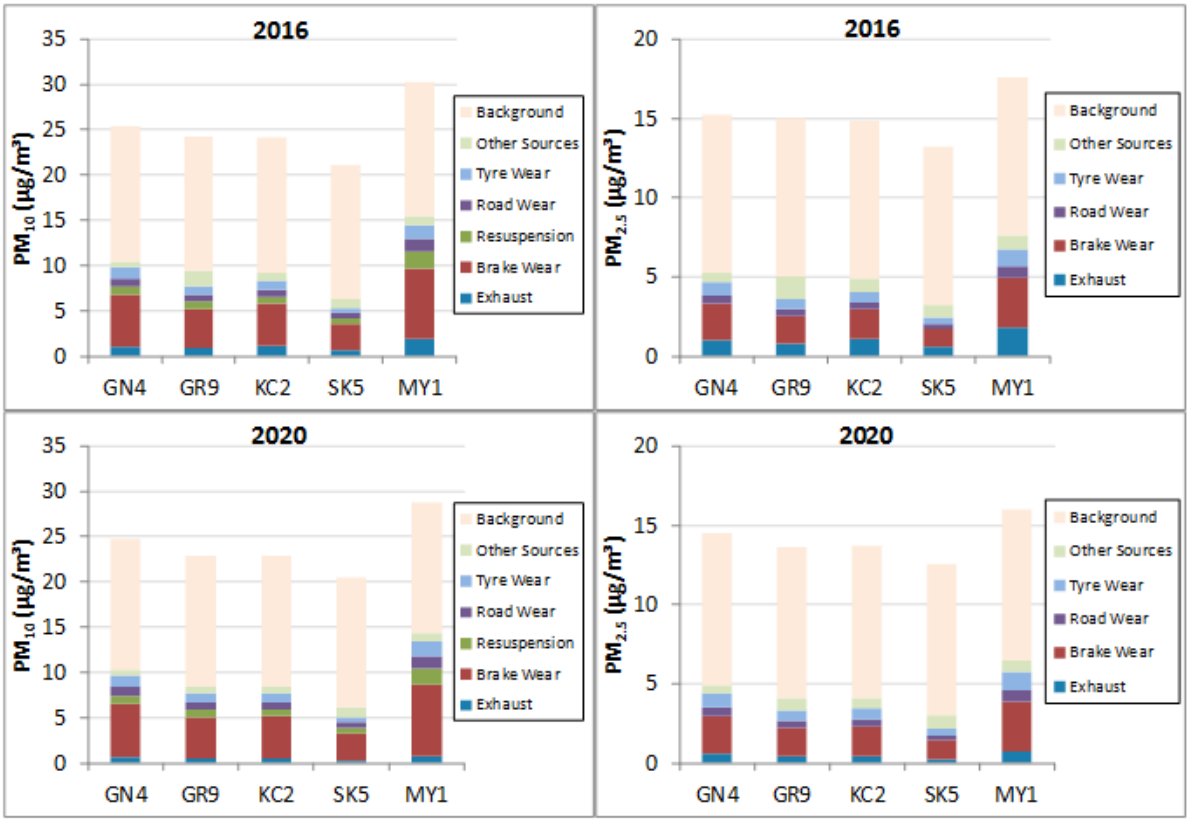

Figure 15: PM_{10} and $PM_{2.5}$ concentrations modelled using ADMS-Urban at five major roads in London apportioned by emission type (µg m^{-3}) for 2016 and 2020. 'Other' represents the contribution of non-traffic sources in the LAEI. The site codes are defined in Table 7.

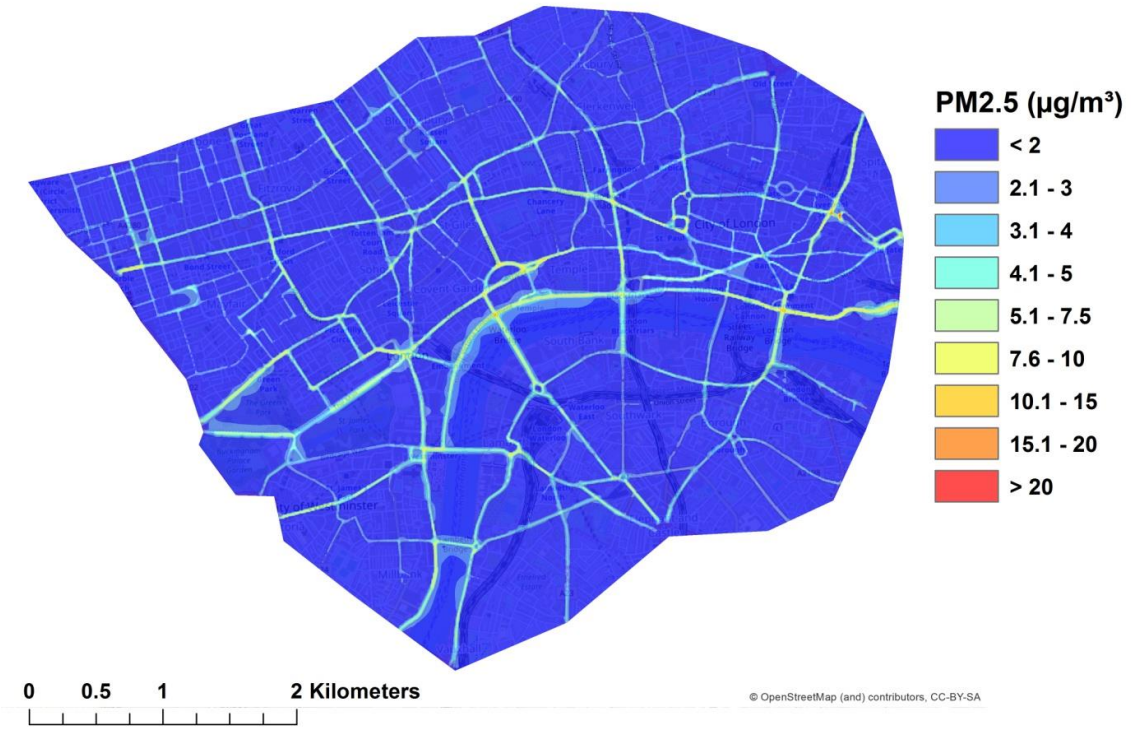

Figure 16: Modelled non-exhaust $PM_{2.5}$ annual average concentration for 2016 within the London Congestion Charging Zone.

4.3 Regional modelling

Most regional modelling studies of PM from transport do not assess the specific impact of non-exhaust emissions. Although emission inventories separate out emissions from tyre and brake wear (NRF 1A3bvi) and road abrasion (NRF 1A3bvii) from exhaust emissions (EMEP/EEA, 2016; See Chapter 2), most studies aggregate these into a single source type: transport emissions (e.g. Archer-Nicholls et al., 2014). Furthermore, the PM are usually then combined with primary PM emissions from other sources into bins of specific sizes. Assumptions are made about the size distribution of the emissions and may not take account of the specific size distribution of non-exhaust PM. Particles within each bin are assumed to be chemically similar (internally mixed), but each bin is chemically distinct from each other (externally mixed). This approach means that the origin of the particles is lost, along with the source specific size or chemical information. Methods for labelling sources of PM have been developed but only applied to broad emission sectors (e.g. transport) (Kranenburg et al, 2013).

In the study by Denier van der Gon et al. (2007) of the contribution of brake wear to atmospheric Cu, the Cu was treated separately from other transport emissions. Cu emissions were included in the LOTOS-EUROS regional chemical transport model by assuming emission factors per kilometre driven by different vehicle types: 3, 6, 10 and 27 mg vkm^{-1} for motorcycles, passenger cars, light duty vehicles and heavy duty vehicles, respectively. Distance driven was by vehicle type was obtained from the Baseline Scenarios for Clean Air for Europe (CAFE) Programme[8]. The Cu particles were assumed to be chemically inert and deposited at a rate based on their size. The total calculated concentration of Cu was dependent on the assumed size fractionation of the particles. Switching from a fine ($PM_{2.5}$) to coarse ($PM_{2.5-10}$) mode ratio of 70:30 to 30:70 lowered the total Cu concentration by around 20%, because coarse mode particles have a higher deposition velocity. Comparing modelled Cu concentrations with observations, Denier van der Gon et al. (2007) estimated that brake wear emissions may be responsible for 50-70% of total Cu emission to air for most of Western Europe, with around 80-90% in the UK.

[8] http://ec.europa.eu/environment/archives/cafe/general/pdf/cafe_lot1.pdf

5 Abatement of Non-Exhaust Emissions of PM

With many factors affecting the formation of non-exhaust particulate emissions it is not surprising that there are many opportunities for particulate emission reduction. Pressure on vehicle manufacturers to reduce non-exhaust particulate matter from an air quality perspective is increasing, but has not yet directly driven technology change or implementation. This section summarises the opportunities presented in the literature, indicating the degree to which currently available technologies can reduce NEE of PM in the areas of brake emissions, tyre emissions and resuspension sources.

5.1 Brake-source particulate matter

Brake system technological development has been driven by commercial interests including friction characteristics, noise and vibration characteristics, wear and durability. Desire to reduce particle emissions from brake systems has arisen from customer drivers such as surface contamination of wheels (e.g. Gaylard, 2010) with only a burgeoning interest in the airborne emission components. As such, there are few studies to date that thoroughly focus on opportunities to reduce particulate emissions through brake system design, material formulation and add-on technologies. This section examines a selection of literature to identify technology options and approaches that are available to reduce brake-source particulate matter from vehicles. As with exhaust particulate matter, care is taken to differentiate between mass-based and number-based measures.

Reduction of brake-source particulate matter can be achieved by:

a) reducing the formation of particles
b) trapping the particles after formation

Altering particulate characteristics after formation (e.g. changing size distribution through enhanced agglomeration) may reduce the negative impact of particles. No information was found in the literature covering this for brake systems so it is not explored further here.

5.1.1 Reducing the formation of brake-source particles

Work showing potential to reduce formation of particulate matter from brakes revolves around new friction material formulations. New formulations have not been driven by quantity of particulate emissions but the materials being emitted. Work related to the Brake Pad Partnership grew out of concerns of Cu emissions in the San Francisco Bay region and has led to a growing interest in reformulating brake pad materials. Particulate emissions more generally, however, do not receive as much attention in the published works in this area.

Perricone et al. (2018), reporting findings from the EU funded REBRAKE project, did focus on airborne particulate emissions. They linked significant ultrafine and fine particle emissions from brake friction surfaces to the evaporation and condensation of binders in the brake materials. The onset of significant particulate emissions was therefore highly temperature dependent. For the sample presented, a 15 °C temperature increase above approximately 170 °C (referred to as a transition temperature) was shown to result in a 5000 times increase in particle number emissions, almost all of which were fine particles and ultrafine particles. Such transition temperatures are material dependent suggesting strong interactions between brake material formulation and particulate emissions. Brake events can give rise to local surface temperatures that exceed such transition temperatures (e.g. Adamowicz and Grzes, 2011) under prolonged, repetitive or extreme braking despite the bulk temperatures potentially remaining lower. This suggests optimization of local material thermal diffusivity could be important for nano-particle reduction and particle number reduction and should be considered closely in new material formulations.

Non-Asbestos Organic (NAO) brake pad formulations (more common in the US and Japanese markets) have demonstrated a 45-48% lower particulate mass emission (Perricone et al., 2018) than Low Metal (LM) content brake pad formulations commonly used in the European market. The costs of NAO brake pads are cited as a barrier to their implementation. Neis et al. (2017) observed NAO pads having larger contact plateau areas than LM pads which may help manage local material temperatures and therefore nano-particle formation from evaporated components such as the softer binders.

Perricone et al. (2018) also showed a heat treatment for the cast iron brake disc resulted in 32% particulate emission reduction by mass without loss in friction performance. This heat treatment effect was attributed to the change in mechanical properties of the disc where its hardness increased from 210 to 473 HB (Brinell Hardness Scale). This work highlighted that the impact of the heat treatment on other brake product commercial requirements was yet to

be evaluated. Matějka et al. (2017) showed that after bedding in of brake pads and discs there is a reduction in the formation of airborne particulates which relates to the creation of the stable friction surface. This suggests caution is needed in evaluating brake source emissions reported at different stages of the component life.

Studies which compare closed and open brake systems such as that by Hagino et al. (2016) show clearly the lower quantity of emitted particles from enclosed (drum brake) systems by approximately a factor of 10. The use of drum brake systems is less prevalent than it has been historically and depends on the vehicle type and market. However, the reduction is attributed to the enclosure of the braking components which retain the particles and therefore enclosure rather than a reversion to brake drum technology is an option provided cost and thermal performance requirements can be met.

It is important that use of new materials in frictional braking systems does not lead to enhanced emissions of known toxic components.

5.1.2 Trapping particles after formation

Commercial interest in filter based abatement of NEE is apparent in patent activity, particularly since the focus of San Francisco authorities on copper emissions from brake materials and subsequently the Brake Pad Partnership (Rosselot, 2006). Little information is available about the performance of the patented systems although the variety in patents published resembles somewhat the technologies explored previously for exhaust particle emission reduction.

Filtration technologies applied in the vicinity of the brake callipers or around the brake disc make use of the induced air flow in the wheel well to pass a fraction of the flow through a filtration element. Fieldhouse and Gelb (2016) reported un-evidenced trapping efficiency of 92% for one such system which could vary the filtered flow depending on thermal dissipation requirements of the brake components. The value of 92% was recorded at the system's highest filtration performance condition and most thermally insulating, thereby being an optimistic assessment. As a mass-based measurement, this is likely to be higher than an equivalent number-based filtration efficiency. They observed a 15-29% reduction in brake wear attributed to reduced surface contamination with wear debris. It is plausible that reduced wear will contribute to reduced particulate emissions for a given material formulation.

Mann-Hummel started trials in 2017 for a variety of filtration solutions for automotive applications including a partial flow fibrous brake dust particle filter. Little information on its

performance was available at the time of preparing this report. Unevidenced marketing information claimed 80% of particles were trapped by the technology. Such filters were reported to be under trial on 10 DHL electric delivery vehicles in five German cities early in 2018. Particle capture systems were also being investigated on the LOWBRASYS Horizon 2020 project due to finish in 2019. Partial flow exhaust flow filtration systems may offer, with caution, an indication of potential. They achieved typically 30-70 % filtration by mass and by number depending on PM loading state and filter design (Schrewe et al, 2012). Such filtration approaches have significant reductions in filtration performance as loading increases as the reduced permeability leads to reduced fractional flow.

Patent activity indicates innovations which may enhance filtration performance including pumped systems, and electro-magnetic enhancements. Performance data are not available in the literature for brake applications of these technologies.

Air filtration systems such as that developed by StreetVac and Mann-Hummel's fine dust particle filter are designed to be fitted in or around vehicles. Induced flows will lead to filtration of wheel well air or surroundings air depending on where it is installed on the vehicle. Filtration performance has not, to our knowledge, been reported. Marketing claims related to Mann-Hummel's fleet refer to emission-neutral vehicles and removal of both brake and tyre particles, both claims unevidenced in the literature at the time of preparing this report.

The potential impact of additional vehicle mass from particle trapping systems on CO_2 emissions needs also to be evaluated.

5.2 Tyre-source particulate matter

Reduction of tyre-source particulate matter can be achieved primarily through reformulation and redesign. Literature does not suggest interest in enclosing the wheel to enable direct capture of tyre source particulates although some approaches to filtering airflows around the tyre were introduced in the previous section.

5.2.1 Reducing particle formation

Modified tyre contact rubber formulations are being explored with the aim of improving the fuel economy – friction performance – wear rate compromises. Polymeric materials are being

explored (e.g. Harper et al., 2017) which can improve the overall compromise and therefore offer the opportunity to reduce wear and potentially particulate emissions. The formulation and design differences between manufacturers and tyre models are substantial causing researchers such as Grigoratos et al. (2018) to conclude that there was no obvious statistical relationship between standard measures of tyre wear and PM or particle number concentrations when comparing five different tyre models. The variation in particle number concentration was between ~2500 and 5200 cm^{-3} showing that there is scope to reduce particle emissions from tyres on the market. The current commercial drivers of fuel economy, friction performance and wear rate may, however, drive reductions in tyre source particulates.

Foitzik et al. (2018) demonstrate the importance of slip angle and longitudinal tyre forces on particle number emissions. This suggests opportunity with increased automation of vehicles in avoiding high emitting conditions. Their data suggested a 0.8-1.8% increase in nanoparticle emissions is plausible for every 10 kg increase in vehicle mass. Light-weighting efforts from manufacturers for improved fuel consumption are therefore contributing to reducing tyre-source particulate matter.

5.3 Resuspended particulate matter

An increment in mineral dust in urban areas and, most especially beside roads, provides evidence of traffic as a source of both road wear and the resuspension of deposited road dust (Amato et al. 2010). From measurements across Spain, Querol et al. (2008) found that the mineral PM_{10} could be divided into three bands depending on location: < 6 µg m^{-3} in rural locations, to 6–8 µg m^{-3} in urban areas and > 8 µg m^{-3} close to roads. In Berlin, Lenshow et al. (2001) estimated that around half of the increment between roadside and background urban locations was due to mineral dust.

The mineral components in PM_{10} vary by location and also by climate, being greater in dryer locations and those near deserts. Road dust is also an important PM_{10} source in cold countries due to the measures to make roads safely drivable in winter. These include both road sanding and the use of studded tyres which wear the road surface. These processes can result in a large surface dust burden remaining on roads during dry days in springtime. Alongside busy roads in Stockholm mean monthly PM_{10} concentrations in springtime were more than 80 µg m^{-3} for March and April (1999–2004) and daily averages reached 200 µg m^{-3} (Norman and Johansson, 2006). In the UK, problems with street dust and resuspension, leading to breaches

of the EU Limit Values for PM_{10}, have been found around construction sites (Fuller and Green, 2004) and on haulage routes to and from waste management sites where the additional PM_{10} load can be greater than up to 33 μg m^{-3} (annual mean) at 15 m from site entrances and 7 μg m^{-3}, 1.1 km from the facility. (Fuller and Baker, 2008; Barratt and Fuller, 2014)

Amato et al. (2010) divides the main control options into three types: street sweeping, street washing and chemical dust suppressants. Here, we also consider best practice measures around construction sites and, where appropriate, focus on UK examples. Road surface texture was shown by China and James (2012) to have a significant effect on resuspension of PM_{10} from soil loading in Las Vegas for mean texture depths of 1 mm or lower. They observed increasing mean texture depth from 0.5 to 1 mm led to a reduction in resuspended PM_{10} mass of ~64%. Although regional effects such as humidity will affect this, it does indicate road surface design as a mechanism for reduction in resuspended particulate emissions.

5.3.1 Street sweeping

Street cleaning has been part of the normal role of local authorities for hundreds of years. This normally comprises the removal of vegetation, dust and rubbish for aesthetics, sanitation and to maintain drainage systems. Mechanical sweepers are increasingly used for this task. These fall into different types that include those that just sweep and those that vacuum the roadway too. Efficiency in removing dust (particle sizes up to 2000 nm) from road surfaces have been found to vary between 5 and 94% dependent on particle size and sweeper technology. Sweepers have been found to be most effective for removing larger visible particles from street surfaces but there is evidence of a threshold effect, with sweepers unable to remove low dust loadings. Better results were obtained where water washing and sweeping were undertaken together (Amato et al., 2010).

Several measurement approaches have been used to assess the effectiveness of road sweeping to control PM_{10} resuspension. These include comparing swept and un-swept stretches of the same road, comparing roadside concentrations to that at a nearby background, looking for changes in emissions ratios before and after sweeping (normally PM_{10} to NO_x) and the use of up and downwind sites paired either side of a road. The majority of eighteen studies reviewed by Amato et al. (2010) did not find an improvement in PM_{10} from the roadway after sweeping. Four studies found an increase in PM_{10} emission factors or concentrations after sweeping. It was suggested that the sweepers moved small particles from the side of the road into the active traffic lanes where they could then be entrained into the air

by passing traffic. However, it was thought possible that, in the longer-term, sweeping might be effective in removing larger dust particles before they are worn down to PM_{10} size fractions.

In the UK, Fuller (2017) studied the impacts of street cleaning in an industrial estate in south London. The estate was home to 150 business and the mixed use of the industrial units brought waste management businesses into conflict with other businesses. PM_{10} concentrations where measured at a fixed point within the estate over a nine-month period. Mean PM_{10} concentrations were 8 µg m^{-3} above that expected on the basis of background sources and primary emissions of exhaust and traffic particles found in typical locations in London. This additional local PM_{10} source comprised 34% of the total measured PM_{10}. It was sensitive to rainfall, decreasing by more than 50% on wet days and those with high relative humidity (<80%). A programme of intensive cleans of the industrial estate were undertaken as a control measure. Both roadways and pavements were swept and washed, vegetation was cleared from pavements and drains unblocked over a four-week period. The intervention led to a downward trend in PM_{10} concentrations, but this was not maintained once the cleaning programme was finished. The most likely explanation being that roadway dust was replenished by fresh material brought out from the waste management sites. The effect of conventional street sweeping could not be detected in the PM_{10} concentrations.

5.3.2 Street washing

Washing is often used as part of street and road cleaning. This can happen on its own or in combination with street sweeping. It would appear that the combination of washing and sweeping is more effective than washing or sweeping alone. Although many studies, including those in Berlin and Bremen did not detect a change (Amato et al., 2010), reductions of around 6% in roadside PM_{10} were found by Norman and Johansson (2006) in springtime in Stockholm on days when street washing took place.

In the UK, Mittal et al. (2013) conducted a short trial in the 300 m long Beech Street road tunnel in the City of London. Beech Street is a bus route, a popular shortcut for London taxis and is also used by used by many pedestrians and cyclists. PM_{10} measurements near the tunnel entrance showed breaches of the EU Limit Value in 2012. Road washing took place on ten nights during spring 2013. Analysis focused on comparing PM_{10} concentrations from inside the tunnel to those measured at an urban background location. Days with road washing were compared to days with no road washing during the trial and also to the pre-trial period. Mean

PM_{10} concentrations from inside the tunnel decreased by a mean of 19 - 25 µg m^{-3} in the six hours following street washing but this benefit was halved by the end of the day.

One of the most detailed studies of pressure washing and sweeping of roads and pavements was undertaken in Barcelona by Amato et al. (2009). Street washing was carried out over ten nights. PM_{10} concentrations were measured at either end of the washed road section, to function as up and downwind sampling points and concentration were also compared to background locations. Roadside PM_{10} concentrations decreased by an average 4–5 µg m^{-3} (7–10%) following the street washing. Concentrations of Cu, Sb, Fe and mineral matter in PM_{10} decreased, indicating the reduced resuspension of material from the roadway but no change was found in elemental carbon which acted as a tracer for exhaust emissions, suggesting that changes in traffic exhaust were not a confounding factor in the analysis.

Amato et al. (2013) identified complex interactions between particles on the road surface and the environment affecting their resuspension. They identified tyre wear as regaining mobility the fastest with other sources such as brake wear, mineral component and exhaust contribution dependent on location. This suggests varied success of sweeping and washing processes are expected depending on location and deposited particulate sources. Resuspension of mineral components, for example, were shown to be somewhat more suppressed by rainwater than tyre wear sources.

5.3.3 Dust suppressants

Dust suppressants are chemicals that are applied to a road surface to reduce resuspension. They vary by chemical composition with the two most tested being calcium magnesium acetate (CMA) and $MgCl_2$. These allow the treated road surface to remain wet at lower levels of relative humidity than an untreated surface. Roadside PM_{10} reductions of 35%, on dry days, were found when CMA was tested as a possible solution to the PM_{10} problems from studded tyres and winter sanding in Stockholm (Norman and Johansson 2006). CMA was also evaluated in Austria as part of the EU Life funded CMA+ project (http://www.life-cma.at). Tests in Klagenfurt, Lienz and Bruneck found that CMA could reduce PM_{10} resuspension by 10 and 20% during the winter months and by up to 40% during summer months allowing the authors to conclude that the product would be effective in controlling both winter dust problems and those around construction sites during the summer. Application of $MgCl_2$ has been mainly tested on paved roads in Norway. In Trondheim, application of a 15% solution of $MgCl_2$ to a highway resulted in an average reduction in the PM_{10} of 17% during dry days. In contrast to these success stories from dust suppressants, studies in German urban areas did not show

any significant reduction in PM_{10} due to CMA use (Amato et al., 2010). A more recent review of dust suppressants (Airuse, 2016) revealed a similar pattern with CMA being effective to address PM_{10} from road dust in Scandinavia but having little impact in Germany or Barcelona (Amato et al., 2014).

CMA has been trialled extensively in London. A small first phase study was undertaken in 2010 and 2011 followed by a more detailed second phase in 2011 and 2012. In phase two CMA was applied to over 30 km of roads and the impacts assessed at nine air quality monitoring sites. A range of data analysis techniques were employed including emissions ratio techniques and statistical approaches that sought to control for meteorological variability. The impact of CMA was not detectable or was not significant along typical London roads. However, CMA did prove effective at locations with local non-exhaust PM_{10} more than 6 µg m^{-3} greater than that expected based on typical NO_X to PM_{10} emissions ratios. In these areas the local non-exhaust PM_{10} decreased by between 12 and 22%, over the study period. The types of location where CMA was effective included proximity to waste management sites, a haulage route adjacent to a large construction site and in a short road tunnel (Barratt et al., 2012).

A more recent study looked at the use of CMA in the yard of a waste management facility and on the public road to and from the facility. PM_{10} measurement was undertaken at three points along the public road, with the furthest being around 450 m from the waste facility entrance, and at nearby background location. Many interventions were tested over the year-long study. Cleaning and sweeping only interventions did not lead to an improvement in PM_{10} near the waste site or along the public road. However, CMA applications reduced the roadside increment in PM_{10} by between 20 and 30% (Mittal and Fuller, 2016).

Dust suppressants are not a permanent treatment for a road surface and need to be reapplied frequently. Studies vary in their findings on reduction of the efficacy of dust suppressants over time. There is evidence that the efficacy may decrease in as little as half a day with no effect being seen after between two and ten days (Airuse, 2016).

5.3.4 Best practice control of resuspended PM_{10} around construction sites

Dust and PM_{10} from construction is acknowledged as an important local source. This has given rise to best practice and London Supplementary planning guidance (SPG) (IAQM, 2014; GLA, 2014). Although this guidance focuses on on-site mitigation the London SPG warns that site track out of material, and therefore locally increased PM_{10} concentrations may occur up to 500

m from large construction sites. The SPG therefore requires developers to reduce this local impact. Suggested control measures include reducing deliveries by road, vehicle wheel washing, road sweeping and washing, and the use of CMA dust suppressants.

Clear lessons from sweeping, washing and dust suppressants shows that they are of variable effectiveness and even under the most optimal conditions dust suppressants may only reduce dust resuspension by around 30 to 40%. The use of dust suppressants around construction and waste sites should therefore be part of wider on-site dust management and is not a substitute for this.

6 Future Trends in Non-Exhaust Emissions of Particulate Matter

The magnitude of non-exhaust emissions of PM as currently estimated has important implications for future PM emissions and air quality, because although current policies on exhaust emissions suggest that emissions of PM per vehicle, both light- and heavy-duty, will decrease significantly, as legislation and policy currently stand this is not necessarily the case for non-exhaust emissions.

Three important issues determine the level and importance of non-exhaust emissions in future years, namely (i) the effect of future vehicle technology, in particular the effect of electric and hybrid vehicles on non-exhaust emissions, (ii) future trends in vehicle activity and (iii) the effect of any future legislation which could affect the level and chemical composition of non-exhaust emissions. These are discussed separately below.

A key feature of UK policy is the encouragement of electric vehicles and the key issue here is how regenerative braking and the changing mass of vehicles will affect NEE. There is considerable uncertainty on this and two points are important. The relative contribution of non-exhaust particle emissions from a battery electric vehicle relative to an equivalent internal combustion engine vehicle is critically dependent upon whether increased road dust resuspension and tyre/road surface wear due to a higher vehicle mass exceed both the amount of exhaust emissions which are no longer present and reductions in brake wear particles due to regenerative braking in electric vehicles. Hall (2017) performed a comparison of braking behaviour between internal-combustion engine vehicles and electric vehicles. He identified not only a reduction in the energy dissipated in the friction brake system from use of regenerative braking, but changes in driving style arising from differences in vehicle behaviour during coast-down events. Brake systems were employed by the driver for less than a quarter of the duration (for an individual event) and less frequently (by up to a factor 8). Energy dissipated in friction brakes was as little as 5% of that of the internal-combustion engine counterpart. The magnitude of these results should be taken with caution due to the limited size of the study, but do suggest further study is warranted. On the other hand, Timmers and Achten (2016) report that the lower energy storage density of electric batteries compared with liquid fuels contribute to a ~24% increase in mass of electric vehicles compared with equivalent conventional vehicles, although mass increases between conventional and hybrid or electric model equivalents can be smaller or greater than this Kollamthodi et al. (2015). This increases the tyre wear, brake wear and resuspended particulate matter emissions leading to

their assessment of only a 1-3% reduction in $PM_{2.5}$ emissions from electric vehicles when compared to conventional powertrains. However, there is a strong incentive for manufacturers to continue to reduce overall electric vehicle mass in order to improve range. Because of this, it would seem likely that the general move to electric vehicles and regenerative braking will lead to an overall reduction in NEE.

A further question arises over autonomous (i.e. self-driving) vehicles. Gawron et al. (2018) have conducted a life cycle assessment of connected and automated vehicles and conclude that the autonomous vehicle will have added mass, electricity demand and aerodynamic drag due to the sensors and computers needed for operation of the vehicle. The increased computing power will have consequences in terms of either installation of greater battery capacity or a reduction in range. However, all of the factors are likely to imply a greater mass for an autonomous battery electric vehicle compared to a conventional human-driven vehicle. Balanced against this, the programming of autonomous battery electric vehicles is likely to determine that only regenerative braking will be used except in emergency stops, and consequently there will be lower brake wear emissions than from a conventional battery electric vehicle. The balance of these factors for emission of non-exhaust particles is likely to prove complex, and no simple statement can be made over the implications of autonomous vehicles for non-exhaust emissions.

Future trends in vehicle activity are crucially important, and some projections for the UK are given in Section 2.1.2. One recent study investigated inter alia these issues (Williams et al., 2018) where the authors assumed that emission factors stay at present levels until 2050 in the absence of any better information. The study investigated two future scenarios produced using the UKTIMES energy system model, consistent with achieving the target for CO_2 equivalent emissions required by the UK Climate Change Act. In both scenarios, vehicle activity was projected to increase substantially from 2010 to 2050, by roughly 50% for cars and heavy goods vehicles and by roughly a factor of 2 for vans. With constant non-exhaust emission factors total UK primary PM_{10} emissions were very similar in 2050 to those in 2010, with reductions in 2050 of only 6-7% on a 2010 emission of 181 kt/yr of PM_{10}. The decrease in exhaust emissions of PM_{10} was offset by the increase in non-exhaust emissions. What this may mean in terms of PM toxicity is not known. While these scenarios are not necessarily accurate forecasts of future activity (the projections of future vehicle activity are particularly uncertain), and emission factors may not remain constant over the period to 2050, the analysis suggests that this issue is of concern and needs more detailed further investigation.

At present there is no legislation covering non-exhaust emissions, nor is there any form of product standard governing the composition of brake systems and tyres, aside from the prohibition on “PAH-rich” extender oils in tyre production as stated in Section 2.1.4. However, methods for the measurement of tyre abrasion rate and of brake-wear emissions are under development, as described in Section 2.3. In the context of the latter, a new brake test cycle has been proposed by Mathissen et al. (2018) which has reinterpreted data collected for fuel economy and exhaust emissions cycles to produce a cycle with more representative brake conditions and therefore be suitable for brake source emissions measurement. The proposed cycle is shown to be more representative of real world driving brake conditions than the exemplar existing brake cycle, which itself was not developed for emission testing. The highly non-linear relationship between brake emissions and brake material temperature makes production of a single representative cycle difficult. The proposed cycle will capture qualitative differences between vehicles and brake systems, however, quantitative predictions of real world brake emissions from cycle test data are likely to remain as elusive as for exhaust emissions. The proposed cycle data presented did not result in brake temperatures exceeding the threshold at which particle number increases significantly and therefore potentially misses the influence of low frequency, more extreme braking events which could contribute significantly to overall particle number. The data presented do not allow a quantitative evaluation of this risk.

The current and projected contributions of non-exhaust emissions to ambient PM concentrations clearly have implications for the achievement of air quality standards and limits in current legislation and for the achievement of WHO Air Quality Guidelines. As well as the legal implications more work needs to be done to assess the potential health impacts of non-exhaust emissions and the need for any composition changes in brakes and tyres. US efforts to reduce copper and other lesser constituents of brake pads by 2021 (EPA 2015, California SB346, 2010) mean alternative material compositions are being sought with the requirement of meeting structural integrity and thermal characteristics, particularly for creating and maintaining a stable friction layer. Straffelini et al. (2015) reviews the substitute options and identifies a number of alternatives including graphite and copper nanoparticles (to reduce the copper content required). Natural substitutes offered mixed braking performance. Subtle changes in ultra-fine and fine particle emissions may arise from new formulations which need careful evaluation with respect to particulate emissions which is not currently a commercial driver.

7 Conclusions and Recommendations

7.1 Conclusions

- Non-exhaust emissions (NEE) of particles apply to all forms of ground transport and can be categorised as those from four sources: brake wear, tyre wear, road surface wear, and resuspended road dust. There may be other sources, e.g. engine belts and clutch plates.

- Quantitative data on the magnitude of non-exhaust emissions are sparse and highly uncertain, particularly when compared to data for exhaust emissions. Emissions vary widely according to brake, tyre and road surface material, and with driving style. As a consequence, emission factors that exist for NEE have a wide range of uncertainty, including wide range in uncertainty in splits between PM_{10}, $PM_{2.5}$ and PM_1 size fractions. Min-max uncertainty ranges spanning a factor of two or more are typical. The exact contribution of non-exhaust emissions to air quality nationally and locally is therefore currently subject to considerable uncertainty.

- The NEE emission factors in current use for the National Atmospheric Emissions Inventory (NAEI) are based on old data (for example, the NEE factors in the EMEP/EEA Air Pollutant Emissions Inventory Guidebook are based on data from the 1990s and the Guidebook has not been updated for 15 years) and have not evolved in time as vehicle designs and vehicle fleet composition have changed, in contrast to the regularly updated emissions factors used for exhaust emissions. The same NEE emission factors are used in many different modelling studies so agreement of outputs across studies does not represent corroboration of accuracy of the input emission factors.

- Acknowledging the considerable uncertainties in the following statements, the UK national emissions inventory indicates that NEE particles from brake wear, tyre wear and road surface wear now constitute the majority source of primary particulate matter (by mass) from road transport in the UK, in both $PM_{2.5}$ and PM_{10} size fractions (60% and 73%, respectively, in 2016). These proportions are set to become even more dominant in the future with continued projected declines in vehicle-fleet exhaust PM emissions. In 2016, NEE particles from brake wear, tyre wear and road surface wear contributed 8.5% and 7.4% of total UK primary PM_{10} and $PM_{2.5}$ emissions, respectively.

- The three sources of NEE in the inventory – brake wear, tyre wear and road surface wear – contribute approximately the same nationally, and are dominantly contributed by cars,

due to the much greater vehicle-km travelled for this class of vehicle. The inventory does not include estimates of road dust resuspension by passing traffic.

- NEE particles are also an important source of metals to the atmosphere – the national emissions inventory estimates contributions of 47% and 21% for Cu and Zn, primarily associated with brake wear and tyre wear, respectively.

- The available data indicate that NEE are especially important in urban environments; the national inventory estimates half of NEE are on urban roads, particularly those associated with braking, owing to the greater braking per km than on non-urban roads. This has implications for the particle mixture to which the population are exposed. However, emissions may also be high in areas such as motorway slip-roads where there has not been as much monitoring activity. Tyre wear emissions are estimated to be greatest on high traffic volume trunk roads and motorways (both urban and rural).

- There is considerable measurement evidence that NEE lead to increased concentrations of PM and some metals at roadside although precise quantification of the NEE contribution is difficult. Data from London Marylebone Road indicate an NEE contribution (including resuspension) of 4-5 $\mu g\ m^{-3}$ to the roadside increment in PM, mostly in the coarse fraction. Other studies, including dispersion modelling, indicate total NEE concentrations including resuspension within PM_{10} of up to several $\mu g\ m^{-3}$ at busy roadside, and in the region 1-2 $\mu g\ m^{-3}$ for urban background in central London.

- At present there is no type approval legislation covering non-exhaust emissions, nor product standards governing the composition of brake systems and tyres (other than for PAH) that are designed explicitly to limit air pollution. Methods of measuring NEE presently lack international consistency. Efforts are ongoing to develop testing approaches that reflect real-world driving conditions. This is most developed at present for measurement of PM and particle number emissions from brake wear. In addition, the European Commission has mandated development of a method for measurement of tyre abrasion rate as part of the Tyre Labelling Regulation. Account must be taken that NEE emissions from different sources are not independent of each other; for example, the extent of tyre wear is dependent on the road surface material.

- Increases in vehicle mass generally increases NEE (but see next point on regenerative braking), which may have implications for electric vehicles, if they are heavier than the conventional diesel and petrol fuelled models they replace because of battery mass, and

to any vehicle with a powertrain that is heavier than the equivalent internal-combustion-engine vehicle. The same applies to autonomous vehicles, which are also heavier than equivalent human-driven vehicles.

- Regenerative braking does not rely on frictional wear of brake materials so vehicles using this braking system totally or partially, for example electric vehicles, should have lower brake wear emissions. However the net balance between reductions in brake wear emissions and potential increases in tyre and road wear emissions and resuspension for vehicles with regenerative braking remains unquantified, and will depend upon road type and driving mode, as both influence the balance between the different sources of emissions. In locations where brake wear makes a major contribution to overall NEE emissions, it seems likely that there will be a net benefit, but this has yet to be demonstrated.

- Mitigation strategies for ambient particle concentrations derived from NEE include the following.

    - The most effective strategies to reduce NEE relate to traffic management: reduce the overall volume of traffic; lower the speed where traffic is free-flowing (such as trunk roads and motorways); and promote driving behaviour that reduces braking and high-speed cornering.

    - Implement regenerative braking, where that does not lead to net disbenefit on road and tyre wear NEE because of increased vehicle mass.

    - Establish particle mass (and/or number) and particle-associated metal emissions limits for brake pad and tyre technologies (including chemical formulation).

    - Trap brake wear particles in the braking system before release into ambient air, although this technology is currently unproven.

    - Reduce the material that is tracked onto public road surfaces as a result of vehicle movements in and out of construction sites, waste-management sites, quarries, farms, and similar.

    - Wash and sweep streets and/or treat street surfaces for dust suppression; it is noted, however, that impacts on airborne PM from trials of these approaches have so far proven inconsistent and any benefits have been short-lived in nature.

- Rail transport (including trams running along urban streets) are also sources of NEE but there has been no quantification of this source locally or nationally in the UK.

7.2 Recommendations

AQEG makes the following scientific recommendations.

- Work towards a consistent approach internationally concerning measurement of non-exhaust emission factors. The emission rates for brake, tyre and road wear will be highly diverse (different materials used, type of road, surface wetness, individual driver braking and cornering habits) so only by making lots of measurements will a robust picture of average and range in NEE emission factors be obtained.

- Understand gains from use of regenerative braking set against potential increased tyre and road wear where vehicles incorporating regenerative braking have increased mass.

- Conduct further studies to quantify the efficacy of technical solutions. Currently, the most promising areas appear to be regenerative braking and variations to formulation of frictional brake components, but research into other braking technologies, including brake wear particle capture, and low-wear tyres, should also be considered.

- Conduct targeted monitoring near areas of high-speed traffic (e.g. motorways) to investigate predicted emissions/concentration hotspots. The best locations for measurements of NEE contributions may not necessarily be coincident with current air quality monitoring sites.

AQEG recommends that policy development with respect to NEE should also consider the following.

- Recognise that NEE are an important source of ambient concentrations of airborne particles, including for vehicles with zero exhaust emissions of particles.

- For the purposes of reducing public exposure to airborne particles, metals and PAHs, NEE should be managed as part of traffic emission policies. An effective tool for NEE abatement is traffic management, specifically reducing the volume, speed and braking intensity of traffic, and increasing the distance between traffic and members of the public.

- In contrast to vehicle exhaust emissions, road-traffic non-exhaust emissions are currently subject to almost no type approvals and regulations.
- The net effect on NEE between reductions associated with regenerative braking and increases associated with increased mass of vehicles with heavier powertrains should be continually re-evaluated as further evidence becomes available.

8 References

Abbasi, S., Jansson, A., Sellgren, U., Olofsson, U., 2013. Particle emissions from rail traffic: A literature review. Critical Reviews in Environmental Science & Technology 43, 2511–2544.

Adamiec, E., Jarosz-Krzeminska, E., Wieszala, R., 2016. Heavy metals from non-exhaust vehicle emissions in urban and motorway road dusts. Environmental Monitoring and Assessment 188, 369.

Adamowicz A., Grzes, P., 2011. Influence of convective cooling on a disc brake temperature distribution during repetitive braking, Applied Thermal Engineering 31, 2177-2185.

Airuse, 2016. The efficacy of dust suppressants to control road dust re-suspension in northern and central Europe. Report: 14. AIRUSE LIFE 11 ENV/ES/584. http://airuse.eu/wp-content/uploads/2013/11/R14_AIRUSE-Dust-suppressants-CNE.pdf

Amato, F., Alastuey, A., Karanasiou, A., Lucarelli, F., Nava, S., Calzolai, G., Severi, M., Becagli, S., Gianelle, V. L., Colombi, C., Alves, C., Custódio, D., Nunes, T., Cerqueira, M., Pio, C., Eleftheriadis, K., Diapouli, E., Reche, C., Minguillón, M. C., Manousakas, M. I., Maggos, T., Vratolis, S., Harrison, R. M., Querol, X., 2016. AIRUSE-LIFE+: a harmonized PM speciation and source apportionment in five Southern European cities. Atmospheric Chemistry and Physics 16, 3289-3309.

Amato, F., Querol, X., Johansson, C., Nagl, C., Alastuey, A., 2010. A review on the effectiveness of street sweeping, washing and dust suppressants as urban PM control methods. Science of the Total Environment 408, 3070-3084.

Amato, F., Cassee, F. R., Denier van der Gon, H., Gehrig, R., Gustafsson, M., Hafner, W., Harrison, R. M., Jozwicka, M., Kelly, F. J., Moreno, T., Prevot, A. S. H., Schaap, M., Sunyer, J., Querol, X., 2014a. Urban air quality: The challenge of traffic non-exhaust emissions. Journal of Hazardous Materials 275, 31-36.

Amato, F., Karanasiou, A., Cordoba, P., Alastuey, A. s., Moreno, T., Lucarelli, F., Nava, S., Calzolai, G., Querol, X., 2014b. Effects of Road Dust Suppressants on PM Levels in a Mediterranean Urban Area. Environmental Science & Technology 48, 8069-8077.

Amato, F., Querol, X., Alastuey, A. S., Pandolfi, M., Moreno, T., Gracia, J., Rodriguez, P., 2009. Evaluating urban PM10 pollution benefit induced by street cleaning activities. Atmospheric Environment 43, 4472-4480.

Amato, F., Schaap, M., Denier van der Gon, H., Pandolfi, M., Alastuey, A. s., Keuken, M., Querol, X., 2013. Short-term variability of mineral dust, metals and carbon emission from road dust resuspension. Atmospheric Environment 74, 134-140.

Archer-Nicholls, S., D. Lowe, S. Utembe, J. Allan, R. A. Zaveri, J. D. Fast, Ø. Hodnebrog, H. Denier van der Gon, and G. McFiggans, 2014, Gaseous chemistry and aerosol mechanism developments for version 3.5.1 of the online regional model, WRF-Chem, Geoscientific Model Development, 7, 2557–2579.

Barlow, T.J., Boulter, P.G., McCrae, I., Sivell, M., Harrison, R., Carruthers, D. and Stocker, J., 2007. Non-exhaust particulate matter emissions from road traffic: summary report. Published project report PPR231.

Barratt, B. M., Fuller, G. W., 2014. Intervention assessments in the control of PM10 emissions from an urban waste transfer station. Environmental Science: Processes & Impacts 16, 1328-1337.

Beddows, D. C. S., Dall'Osto, M., Olatunbosun, O. A., Harrison, R. M., 2016. Detection of brake wear aerosols by aerosol time-of-flight mass spectrometry. Atmospheric Environment 129, 167-175.

Beddows, D. C. S., Harrison, R. M., Green, D. C., Fuller, G. W., 2015. Receptor modelling of both particle composition and size distribution from a background site in London, UK. Atmospheric Chemistry and Physics 15, 10107-10125.

Boucher, J., Friot D., 2017. Primary microplastics in the oceans: A global evaluation of sources. International Union for Conservation of Nature and Natural Resources. Gland, Switzerland.

Brookes et al. 2017, Technical report on UK supplementary assessment under The Air Quality Directive (2008/50/EC), The Air Quality Framework Directive (96/62/EC) and Fourth Daughter Directive (2004/107/EC) for 2015. Report for DEFRA and devolved administrations.

California Senate Bill No 346, 2010. “SB-346 Hazardous materials: motor vehicle brake friction materials”

China, S., James, D. E., 2012. Influence of pavement macrotexture on PM10 emissions from paved roads: A controlled study. Atmospheric Environment 63, 313-326.

COMEAP, 2010. The mortality effects of long-term exposure to particulate air pollution in the United Kingdom, UK Department of Health Committee on the Medical Effects of Air Pollutants. ISBN 978-0-85951-685-3, http://comeap.org.uk/documents/reports.html.

COMEAP, 2015. Statement on the evidence for differential health effects of particulate matter according to source or components, UK Department of Health Committee on the Medical Effects of Air Pollutants, https://www.gov.uk/government/publications/particulate-air-pollution-health-effects-of-exposure.

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

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

Defra (2018). UK Informative Inventory Report (1990 to 2016). https://uk-air.defra.gov.uk/assets/documents/reports/cat07/1803161032_GB_IIR_2018_v1.2.pdf

de la Paz, D., Borge, R., Vedrenne, M., Lumbreras, J., Amato, F., Karanasiou, A., Boldo, E. and Moreno, T., 2015. Implementation of road dust resuspension in air quality simulations of particulate matter in Madrid (Spain), Frontiers in Environmental Science 3, 72.

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

Denier van der Gon, H., Hulskotte, J., Jozwicka, M., Kranenburg, R., Kuenen, J. and Visschedijk, A., 2018. Chapter 5 - European Emission Inventories and Projections for Road Transport Non-Exhaust Emissions: Analysis of Consistency and Gaps in Emission Inventories From EU Member States. In: Non-Exhaust Emissions, (ed. Amato, F.), Academic Press, 101-121. doi.org/10.1016/B978-0-12-811770-5.00005-4.

EIONET, 2018. EIONET Central Data Repository. UK emission projections available at http://cdr.eionet.europa.eu/gb/un/clrtap/projected/envwqfz4g/Annex_IV_Projections_GB_2018_v1_0.xls/manage_document

EMEP/EEA, 2009. EMEP/EEA Air pollutant emission inventory guidebook 2009. EEA, Copenhagen.

EMEP/EEA, 2013. EMEP/EEA Air pollutant emission inventory guidebook 2013. Part B. 1.A.3.b.vi-vii Road vehicle tyre and brake wear, road surface wear.

EMEP/EEA, 2016. EMEP/EEA Air Pollutant Emission Inventory Guidebook 2016. EEA Report No 21/2016. Road Transport Tyre and Brake Wear and Road Abrasion. 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

EPA, 2015. “Memorandum of understanding on copper mitigation in watersheds and waterways between U.S. Environmental Protection Agency and …” January 2015

Fabretti, J. F., Sauret, N., Gal, J. F., Maria, P. C., Schärar, U., 2009. Elemental characterization and source identification of PM2.5 using Positive Matrix Factorization: The Malraux road tunnel, Nice, France. Atmospheric Research 94, 320-329.

Fieldhouse, J. D., Gelb, J., 2016. New Developments of an On-Vehicle Brake Pad Waste Collection System. SAE International Journal of Passenger Cars - Mechanical Systems 9, 1245-1254.

Foitzik, M. J., Unrau, H. J., Gauterin, F., Dörnhöfer, J., Koch, T., 2018. Investigation of ultra fine particulate matter emission of rubber tires. Wear 394-395, 87-95.

Fuller, G.W. 2017. “Willow Lane Industrial Estate Analysis of PM10 measurements.” King’s College London.

Fuller, G. W., Baker, T. D., 2008. “PM10 source apportionment at Bexley 4, Manor Road, Erith.” King’s College London.

Fuller, G., Baker, T., Tremper, A., Green, D., Priestman, M., Carslaw, D., Dajnak, D., Beevers, S., 2014. Air pollution emissions from diesel trains in London. King’s College London.

Fuller, G. W., Green, D., 2004. The impact of local fugitive PM10 from building works and road works on the assessment of the European Union Limit Value. Atmospheric Environment 38, 4993-5002.

Furger, M., Minguillón, M. C., Yadav, V., Slowik, J. G., Hüglin, C., Fröhlich, R., Petterson, K., Baltensperger, U., Prévôt, A. S. H., 2017. Elemental composition of ambient aerosols measured with high temporal resolution using an online XRF spectrometer. Atmospheric Measurement Techniques 10, 2061-2076.

Gawron, J. H., Keoleian, G. A., De Kleine, R. D., Wallington, T. J. and Kim, H. C., 2018. Life Cycle Assessment of Connected and Automated Vehicles: Sensing and Computing Subsystem and Vehicle Level Effects, Environmental Science & Technology 52, 3249-3256.

Gaylard A P, Lynch D, Amodeo J and Amunugama R, 2010. “The Simulation of Brake Dust Deposition” presented at the 8th MIRA International Vehicle Aerodynamics Conference, 13-14 October 2010.

Gehrig, R., Hill, M., Lienemann, P., Zwicky, C.N., Bukowiecki, N., Weingartner, E., Baltensperger, U., Buchmann, B., 2007. Contribution of railway traffic to local PM_{10} concentrations in Switzerland. Atmospheric Environment 41, 923–933.

Gietl, J.K., Lawrence, R., Thorpe, A.J., Harrison R.M., 2010. Identification of brake wear particles and derivation of a quantitative tracer for brake dust at a major road, Atmospheric Environment, 44, 141-146.

Greater London Authority (GLA), 2014. “The control of dust and emissions during construction and demolition: supplementary planning guidance.” Greater London Authority, London.

Grigoratos, T., Gustafsson, M., Eriksson, O., Martini, G., 2018. Experimental investigation of tread wear and particle emission from tyres with different treadwear marking. Atmospheric Environment 182, 200-212.

Grigoratos, T., Martini, G., 2015. Brake wear particle emissions: a review. Environmental Science and Pollution Research 22, 2491-2504.

Gustafsson, M. (2018) Chapter 8 - Review of Road Wear Emissions: A Review of Road Emission Measurement Studies: Identification of Gaps and Future Needs. In: Non-Exhaust Emissions, (ed. Amato, F.), Academic Press, 161-181. doi.org/10.1016/B978-0-12-811770-5.00008-X.

Hagino, H., Oyama, M., Sasaki, S., 2016. Laboratory testing of airborne brake wear particle emissions using a dynamometer system under urban city driving cycles. Atmospheric Environment 131, 269-278.

Hall, T. J., 2017. A Comparison of Braking Behavior between an IC Engine and Pure Electric Vehicle in Los Angeles City Driving Conditions, SAE Technical Paper 2017-01-2518.

Harper, M., Tardiff, J., Haakenson, D., Joandrea, M., Knych, M., 2017. Tire Tread Performance Modification Utilizing Polymeric Additives. SAE International Journal of Vehicle Dynamics, Stability, and NVH 1, 179-189, doi:10.4271/2017-01-1502.

Harrison, R.M., Beddows, D.C.S., Dall'Osto, M., 2011. PMF analysis of wide-range particle size spectra collected on a major highway, Environmental Science & Technology, 45, 5522-5528.

Harrison, R. M., Jones, A. M., Gietl, J., Yin, J., Green, D. C., 2012. Estimation of the contributions of brake dust, tire wear, and resuspension to nonexhaust traffic particles derived from atmospheric measurements. Environmental Science & Technology 46, 6523-6529.

Harrison, R. M., Yin, J. X., Mark, D., Stedman, J. R., Appleby, R. S., Booker, J. and Moorcroft, S., 2001. Studies of the coarse particle (2.5-10 μm) component in UK urban atmospheres, Atmospheric Environment 35, 3667-3679.

Hartmann, N., Hüffer, T., Thompson, R.C., Hassellöv, M., Verschoor, A., Daugaard, A.E., Rist, S., Karlsson, T.M., Brennholt, N., Cole, M., Herrling, M.P., 2019. Are we speaking the same language? Recommendations for a definition and categorization framework for plastic debris. Environmental Science & Technology 53, 1039–1047.

Hertel, O., Berkowicz, R. and Larssen, S., 1990, 'The Operational Street Pollution Model (OSPM).' *18th International meeting of NATO/CCMS on Air Pollution Modelling and its* Applications. Vancouver, Canada, pp741-749.

Hood C, Carruthers D, Seaton M, Stocker J and Johnson K, 2014. *Urban canopy flow field and advanced street canyon modelling in ADMS-Urban.*16th International Conference on Harmonisation within Atmospheric Dispersion Modelling for Regulatory Purposes, Varna, Bulgaria, September 2014.

Hood, C., MacKenzie, I., Stocker, J., Johnson, K., Carruthers, D., Vieno, M., Doherty, R., 2018. Air quality simulations for London using a coupled regional-to-local modelling system. Atmospheric Chemistry and Physics 18, 11221-11245.

Hopke, P. K.; Harrison, R. M.; De Leeuw, F.; Querol, X., Current State of Particulate Air Quality. In Non-Exhaust Emission: An Urban Air Quality Problem for Public Health: Impact and mitigation measures, pp 1-19, Amato, F., Ed. Academic Press: London, 2018.

Hulskotte, J. H. J., Roskam, G. D., Denier van der Gon, H. A. C., 2014. Elemental composition of current automotive braking materials and derived air emission factors. Atmospheric Environment 99, 436-445.

Institute of Air Quality Management (IAQM), 2014. “Guidance on the assessment of dust from demolition and construction.” IAQM, London.

Klein, J., Hulskotte J., Ligterink, N., Dellaert, S., Molnár, H., Geilenkirchen, G., 2018. Methods for calculating the emissions of transport in the Netherlands. https://english.rvo.nl/sites/default/files/2018/04/Klein-et-al-2018-%20Methods-for-calculating-emissions-transport-Netherlands-2018.pdf

Klimont, Z., Cofala, J., Bertok, I., Amann, M., Heyes, C., Gyarfas, F., 2002. Modelling particulate emissions in Europe - a framework to estimate reduction potential and control costs. Interim Report IR-02-076, 109

Kole, P.J., Löhr, A.J., Van Belleghem, F., Ragas, A., 2017. Wear and tear of tyres: a stealthy source of microplastics in the environment. International Journal of Environmental Research and Public Health, 14, 1265.

Kollamthodi, S., Kay, D., Skinner, I, Dun, C, Hausberger, S., 2015. The potential for mass reduction of passenger cars and light commercial vehicles in relation to future CO_2 regulatory requirements, Report for the European Commission – DG Climate Action, Ref. CLIMA.C.2/FRA/2013/0006/SR1, Ricardo AEA report ED58751. https://ec.europa.eu/clima/sites/clima/files/transport/vehicles/docs/ldv_downweighting_co2_report_en.pdf

Kouridis, C., Gkatzoflias, D., Kioutsioukis, I., Ntziachristos, L., Pastorello, C., Dilara, P., 2010. Uncertainty Estimates and Guidance for Road Transport Emission Calculations. Joint Research Centre report no. JRC 57352, doi:10.2788/78236.

Kranenburg, R., Segers, A. J., Hendriks, C., Schaap, M., 2013. Source apportionment using LOTOS-EUROS: module description and evaluation. Geoscientific Model Development 6, 721-733.

Kreider, M.L., Panko, J.M., McAtee, B.L., Sweet, L.I., Finley, B.L., 2010. Physical and chemical characterization of tire-related particles: Comparison of particles generated using different methodologies, Science of the Total Environment, 408, 652-659.

Lenschow P., Abraham H.J., Kutzner K., Lutz M., Preuß J.D., Reichenbächer W., 2001. Some ideas about the sources of PM10. Atmospheric Environment, 35, Supplement 1, S23-S33.

London Atmospheric Emissions Inventory (LAEI) 2010 Methodology Document, https://data.london.gov.uk/dataset/london-atmospheric-emissions-inventory-2010

Lough, G. C., Schauer, J. J., Park, J. S., Shafer, M. M., DeMinter, J. T., Weinstein, J. P., 2005. Emissions of Metals Associated with Motor Vehicle Roadways. Environmental Science & Technology 39, 826-836.

Matejka, V., Metinöz, I., Wahlström, J., Alemani, M., Perricone, G., 2017. On the running-in of brake pads and discs for dyno bench tests. Tribology International 115, 424-431.

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

Mittal, L., Barratt, B., Fuller, G.W., 2013. “Effects of street washing on PM_{10} concentrations at Beech Street Tunnel.” King’s College London.

Mittal, L., Fuller, G.W., 2016. “Study to assess the effectiveness of CMA dust suppressant and deep cleaning in reducing PM_{10} concentrations in the Scrubs Lane industrial area.” King’s College London.

NAEI, 2018. UK National Atmospheric Emissions Inventory, http://naei.beis.gov.uk/

NECD, 2016. See https://www.eea.europa.eu/themes/air/national-emission-ceilings/national-emission-ceilings-directive

Neis, P. D., Ferreira, N. F., Fekete, G., Matozo, L. T., Masotti, D., 2017. Towards a better understanding of the structures existing on the surface of brake pads. Tribology International 105, 135-147.

Nemitz, E., Fowler, D., Dorsey, J. R., Theobald, M. R., McDonald, A. D., Bower, K. N., Beswick, K. M., Williams, P. I. and Gallagher, M. W., 2000. Direct measurements of size-segregated particle fluxes above a city, Journal of Aerosol Science, 31, 116-117.

Norman, M., Johansson, C., 2006. Studies of some measures to reduce road dust emissions from paved roads in Scandinavia. Atmospheric Environment 40, 6154-6164.

Padoan, E., Amato, F., 2018. Vehicle Non-Exhaust Emissions: impact on Air Quality. In: Non-Exhaust Emission - An Urban Air Quality Problem for Public Health: Impact and mitigation measures, pp 21-65, Amato, F., Ed. Academic Press: London.

Padoan, E., Ajmone-Marsan, F., Querol, X. and Amato, F., 2018. An empirical model to predict road dust emissions based on pavement and traffic characteristics, Environmental Pollution 237, 713-720.

Panko, J.M., Chu, J., Kreider, M.L., Unice, K.M., 2013. Measurement of airborne concentrations of tire and road wear particles in urban and rural areas of France, Japan, and the United States. Atmospheric Environment, 72, 192-199.

Panko, J.M., Hitchcock, K.M., Fuller, G.W., Green, D., 2019. Evaluation of tire wear contribution to $PM_{2.5}$ in urban environments. Atmosphere 10, 99.

Pant, P., Harrison, R. M., 2013. Estimation of the contribution of road traffic emissions to particulate matter concentrations from field measurements: A review. Atmospheric Environment 77, 78-97.

Perricone, G., Matejka, V., Alemani, M., Valota, G., Bonfanti, A., Ciotti, A., Olofsson, U., Söderberg, A., Wahlström, J., Nosko, O., Straffelini, G., Gialanella, S., Ibrahim, M., 2018. A concept for reducing PM_{10} emissions for car brakes by 50%. Wear 396-397, 135-145.

Querol, X., Alastuey, A., Moreno, T., Viana, M.M., Castillo, S., Pey, J., Rodríguez, S., Artiñano, B., Salvador, P., Sánchez, M. and Dos Santos, S.G., 2008. Spatial and temporal variations in airborne particulate matter (PM_{10} and $PM_{2.5}$) across Spain 1999–2005, Atmospheric Environment 42, 3964-3979.

Rosselot, K. S., 2006. “Copper released from brake lining wear in the San Francisco Bay area”, prepared for the Brake Pad Partnership, January 2006, https://suscon.org/wp-content/uploads/2016/12/BrakeSourcesReportFinal01-30-06a.pdf, accessed 29 Jun 2018.

RWS (2008), Remslijtage, Versie juni 2008, Rijkswaterstaat – Waterdienst in samenwerking met DELTARES en TNO, as cited in Klein et al. (2018).

Schrewe, K., Steigert, S., Neumann, P., 2012. Partial Flow Sintered Metal Filter as Part of a Tier 4 Emission Reduction Strategy. SAE International Journal of Fuels and Lubricants 5, 1291-1299, https://doi.org/10.4271/2012-01-1731.

Sternbeck, J., Sjodin, A., Andreasson, K., 2002. Metal emissions from road traffic and the influence of resuspension - results from two tunnel studies. Atmospheric Environment 36, 4735-4744.

Straffelini, G., Ciudin, R., Ciotti, A., Gialanella, S., 2015. Present knowledge and perspectives on the role of copper in brake materials and related environmental issues: A critical assessment. Environmental Pollution 207, 211-219.

Thorpe, A., Harrison, R. M., 2008. Sources and properties of non-exhaust particulate matter from road traffic: A review. Science of the Total Environment 400, 270-282.

Ten Broeke, H., Hulskotte, J. & Denier van der Gron, H., 2008. Emissies door bandenslijtage afkomstig van het wegverkeer, Rijkswaterstaat Waterdienst, as cited in Klein et al. (2018).

Timmers, V. R. J. H., Achten, P. A. J., 2016. Non-exhaust PM emissions from electric vehicles. Atmospheric Environment 134, 10-17.

TNO, 1997. Particulate matter emissions (PM_{10}, $PM_{2.5}$, $PM_{0.1}$) in Europe in 1990 and 1993. TNO Institute of Environmental Sciences, Energy Research and Process Innovation, Apeldoorn, the Netherlands.

USEPA, 1995. Compilation of air pollutant emission factors, USEPA Report AP-42, Volume I, 5th edition.

Venkatram, A., 2000. A critique of empirical emission factor models: a case study of the AP-42 model for estimating PM_{10} emissions from paved roads, Atmospheric Environment 34, 1-11.

VDA, 2017. The Contribution of Brake Wear Emissions to Particulate Matter in Ambient Air. Ricardo Energy & Environment Report FAT 301 for the Verband der Automobilindustrie, https://www.vda.de/de/services/Publikationen/fat-schriftenreihe-301.html

Visser, S., Slowik, J. G., Furger, M., Zotter, P., Bukowiecki, N., Canonaco, F., Flechsig, U., Appel, K., Green, D. C., Tremper, A. H., Young, D. E., Williams, P. I., Allan, J. D., Coe, H.,

Williams, L. R., Mohr, C., Xu, L., Ng, N. L., Nemitz, E., Barlow, J. F., Halios, C. H., Fleming, Z. L., Baltensperger, U., Prévôt, A. S. H., 2015. Advanced source apportionment of size-resolved trace elements at multiple sites in London during winter. Atmospheric Chemistry and Physics 15, 11291-11309.

Vogt, M., Nilsson, E. D., Ahlm, L., Mårtensson, E. M., Johansson, C., 2011. The relationship between 0.25-2.5 mm aerosol and CO2 emissions over a city. Atmospheric Chemistry and Physics 11, 4851-4859.

Vogt, M., Nilsson, E. D., Ahlm, L., Mårtensson, E. M., Johansson, C., 2011. Seasonal and diurnal cycles of 0.25-2.5 mm aerosol fluxes over urban Stockholm, Sweden. Tellus B: Chemical and Physical Meteorology 63, 935-951.

Weckwerth, G., 2001. Verification of traffic emitted aerosol components in the ambient air of Cologne (Germany). Atmospheric Environment 35, 5525-5536.

Williams, M. L., Lott, M. C., Kitwiroon, N., Dajnak, D., Walton, H., Holland, M., Pye, S., Fecht, D., Toledano, M. B. and Beevers, S. D., 2018. The Lancet Countdown on health benefits from the UK Climate Change Act: a modelling study for Great Britain, The Lancet Planetary Health 2, e202-e213.

Winther, M., Slentø, E., 2010: Heavy Metal Emissions for Danish Road Transport. National Environmental Research Institute, Aarhus University, Denmark. 99 pp. – NERI Technical Report no. 780. Available at: http://www.dmu.dk/Pub/FR780.pdf

9 Appendices

9.1 Non-exhaust emissions of PM and metals from railways

The first commercial rail transport appeared in the UK between 1804 and 1812 and the first underground railway opened in 1893, around ten years before the invention of the four-stroke gasoline engine. Despite railways pre-dating motorised road traffic, there have been relatively few studies of the non-exhaust emission from rail transport and, like road transport, there are no legislation or regulations to control emissions or concentrations (Abbasi et al. (2012).

The early investigations of wear emissions from trains noted high emissions and concentrations in underground railways as early as 1909. As a consequence, studies of non-exhaust emissions from railways have mainly focused on underground railways with London and Beijing being amongst the thirteen cities reported by Abbasi et al. (2012).

One of the most extensive investigations of ambient non-exhaust particles from railways was undertaken by Gehrig et al. (2007) who made measurements at various distances from an electrified rail line in Switzerland. At around 10 m from the trackside, PM_{10} concentrations were found to be around 1 µg m^{-3} above that measured at a nearby background site. This was dominated by Fe with smaller contributions from Cu, Mn and Cr. The Fe particles were predominately (72%) in the coarse particle size. Particle concentrations reduced with distance from the railway line; PM_{10} concentrations at 120 m from the railway were only 25% of that measured at 10 m.

In London, Fuller et al. (2014) measured the metallic composition of PM_{10} at the boundary fence alongside the Paddington and East Coast mainlines (Southall and near Highbury (Arsenal). It was difficult to isolate sources of metallic PM from the railway from those from other urban sources such as traffic. Ambient emissions ratios of black carbon and Fe were derived from traffic sources in London and used to separate the Fe of road traffic origin from the Fe from the railway. Using this approach Fe from rail wear was estimated to be between 0.8 ± 0.5 µg m^{-3} and 1.2 ± 0.8 µg m^{-3} as a contribution to PM_{10}, if all Fe was present as Fe_2O_3.

Abbasi et al. (2012) describes how emissions are determined by a range of operational factors including axle load, bogie design, wheel and brake materials, and braking systems. Various control measures have been suggested to control non-exhaust PM from railways. These include better track layouts, optimising train wheel profiles and applying friction modifiers. Improved bogie designs have also been suggested including articulated bogies, active wheel steering and better suspension. Radial grooves in brake discs have been found to reduce

brake wear debris and choice of brake pad can decrease emissions of some metal particles. Brake wear can also be reduced by electric or regenerative brakes that are becoming more commonplace on commuter trains.

9.2 Non-UK national inventory estimates of non-exhaust emissions from railways

There are no requirements for including non-exhaust emissions from railways in national inventories reported under the NECD and UNECE CLRTAP, nor are there any recommended emission factors and methodologies given in the EMEP/EEA Emissions Inventory Guidebook. However, some countries, but not the UK, have included some estimates for their inventories.

9.2.1 Netherlands

The Netherlands is the country that has given most attention to this source of emissions. This is evident from the Netherlands national inventory report, but also, as part of its sustainability programme, the Dutch state-owned passenger rail company (NS) has been particularly interested in the contribution of its rail operations to PM from its mainly electric trains.

The following text is taken from the Netherlands national inventory report produced by RIVM: *"PM_{10} emissions due to wear on overhead contact lines and carbon brushes from railways are calculated using a study conducted by NS-CTO (1992) on the wear on overhead contact lines and the carbon brushes of the collectors on electric trains. For trams and metros, the wear on the overhead contact lines has been assumed to be identical to that on railways. The wear on current collectors has not been included, because no information was available on this topic. Carbon brushes, besides copper, contain 10% lead and 65% carbon. Based on the NS-CTO study, the percentage of particulate matter in the total quantity of wear debris was estimated to be 20%. Because of their low mass, these particles probably remain airborne. It is estimated that approximately 65% of the wear debris ends up in the immediate vicinity of the railway, while 5% enters the ditches alongside the railway line (Coenen & Hulskotte, 1998). According to the NS-CTO study, the remainder of the wear debris (10%) does not enter the environment, but attaches itself to the train surface and is captured in the train washing facilities."*

It is not thought that further research into this area has been done since the 1990s.

No emission factors or emissions are given for railway NEE in the Dutch inventory report. The overall contribution of the rail sector to $PM_{2.5}$ emissions in 2016 was 0.5% but that would include exhaust emissions from diesel freight trains. The metals which had the highest contribution from the rail sector overall were Cu at 14% and Pb at 2.9%; for all other metals the contributions were less than 0.1%.

9.2.2 Germany

Germany also includes estimates of NEE from the rail sector. The German inventory information is provided on a wiki at https://iir-de.wikidot.com/1-a-3-c-railways. Electricity is responsible for 80% of all railway traction power.

Germany includes emissions from the contact line, braking and from the wheel/track interface. The report states that emission factors are calculated from PM_{10} emission estimates directly provided by the German railway company Deutsche Bahn AG. Emission factors for emissions of Cu, Ni and Cr are calculated via typical shares of the named metals in the contact line (Cu) and in the braking systems (Ni and Cr). The factors for wheel/track are given as 0.018 gPM_{10}/km and 0.009 $gPM_{2.5}$/km. Emission factors for the other NEE sources are shown below.

Table 8: Newly implemented emission factors, in g/km

	PM_{10}[1]	$PM_{2.5}$[2]	TSP[3]	Cr[4]	Ni[5]	Cu[6]
wear of braking system	0.0060	0.0030	0.0060	0.00006	0.00012	NA
wear of contact line	0.0003	0.0001	0.0003	NA	NA	0.0003

[1] EF(PM_{10}) estimated from original PM_{10} emissions provided from the German railroad company Deutsche Bahn AG and transport performance data available from TREMOD-

[2] EF($PM_{2.5}$) estimated as a fraction of 50% of the EF(PM_{10})

[3] EF(TSP) similar to the EF(PM_{10})

[4] EF(Cr) estimated via an assumed chromium content in steel used for braking systems of 1 per cent

[5] EF(Ni) estimated via an assumed nickel content in steel used for braking systems of 2 per cent

[6] EF(Cu) estimated via an assumed copper content in the contact line of 100 per cent

The German inventory for 2016 implies that the rail sector, as a whole, contributes around 4% of the total national emissions of PM_{10} and $PM_{2.5}$. Whilst this would include emissions from diesel trains, a chart on the wiki suggests that >90% of this comes from these NEE sources, a lot higher than implied by the Dutch inventory. The overall contribution of the rail sector to total metal emissions in Germany is 31% for Cr, 35% for Ni and 4% for Cu.

9.2.3 France

The national inventory report for France makes reference to NEE from railways. From the information given in gTSP/km for brake, wheel and contact line sources, the following emission factors can be derived, but it seems highly likely that the units given in the report should be in mg/km rather than the g/km stated. The factors are stated to derive from a pers comm in 2002 and a German and CITEPA report dated 2002-2005.

	PM_{10}	$PM_{2.5}$
	g/km	g/km
Brake	5.0	2.3
Wheel/track	3.4	1.0
Contract line	0.16	0.02

The French inventory for 2016 implies that, overall, railways contribute 0.8% to total PM_{10} emissions in France, 0.5% for $PM_{2.5}$ but 23% of Cu emissions. These will include exhaust emissions from diesel trains.