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A Review of Road Traffic-Derived Non-Exhaust Particles: Emissions, Physicochemical Characteristics, Health Risks, and Mitigation Measures

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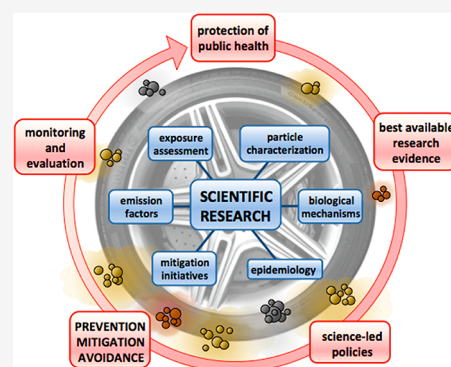
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ABSTRACT: Implementation of regulatory standards has reduced exhaust emissions of particulate matter from road traffic substantially in the developed world. However, nonexhaust particle emissions arising from the wear of brakes, tires, and the road surface, together with the resuspension of road dust, are unregulated and exceed exhaust emissions in many jurisdictions. While knowledge of the sources of nonexhaust particles is fairly good, source-specific measurements of airborne concentrations are few, and studies of the toxicology and epidemiology do not give a clear picture of the health risk posed. This paper reviews the current state of knowledge, with a strong focus on health-related research, highlighting areas where further research is an essential prerequisite for developing focused policy responses to nonexhaust particles.



KEYWORDS: exposure assessment, health effects, mitigation, nonexhaust emissions, road traffic, toxicity

1. INTRODUCTION

A compelling body of evidence exists associating air pollution with increased morbidity and mortality from cardiorespiratory disease and lung cancer, with further accumulating evidence for diabetes, neurological damage, and adverse birth outcomes.¹ The significance of the problem is reflected in the updated World Health Organization Global Air Quality Guidelines, which recently recommended substantially lower air quality limits for PM_{2.5}, PM₁₀ (particulate matter less than 2.5 and 10 μm in diameter respectively), and nitrogen dioxide.² Increasing urbanization brings an ever-growing exposure to air pollutants from road traffic that is linked to acute and chronic health effects.³ Mechanistic studies in humans, animals and cells support epidemiological findings,⁴ demonstrating biological plausibility for multiorgan effects through an array of biological mechanisms.^{5–8}

Traffic-derived air pollution comprises a mixture of gaseous pollutants and PM from fuel combustion and lubricant volatilization in exhaust (tailpipe) emissions. Road transport is also a source of nonexhaust emissions (NEE). These comprise of particles from mechanical abrasion of brakes and tires, erosion of road surfaces and resuspension of a mixture of dust that accumulates on road surfaces, and volatile organic compounds from evaporative loss of fuels and release of solvents.^{9,10} Most epidemiological and experimental research

into traffic-related pollution focuses on particulate and gaseous pollutants emitted from the exhaust, particularly from diesel-fueled vehicles. In contrast, particulates from NEE have been woefully understudied. NEE, especially those from brake and tire wear, are an important source of metals in urban atmospheres.¹¹ The UK emissions inventory estimates mass contributions of 47% and 21% to national total airborne emissions of Cu and Zn, respectively.¹² These metals, as well as those such as Fe from brake wear, catalyze the formation of reactive oxygen species (ROS) in the respiratory tract lining fluids, challenging antioxidants and metal-binding proteins that protect the epithelial surface of the lung.¹³ While invariably being associated with coarse-mode PM, a considerable fraction of abrasion-derived particles exist within the fine and ultrafine fractions,^{14,15} engendering NEE with a high capacity for harm owing to a larger reactive surface area and the ability to

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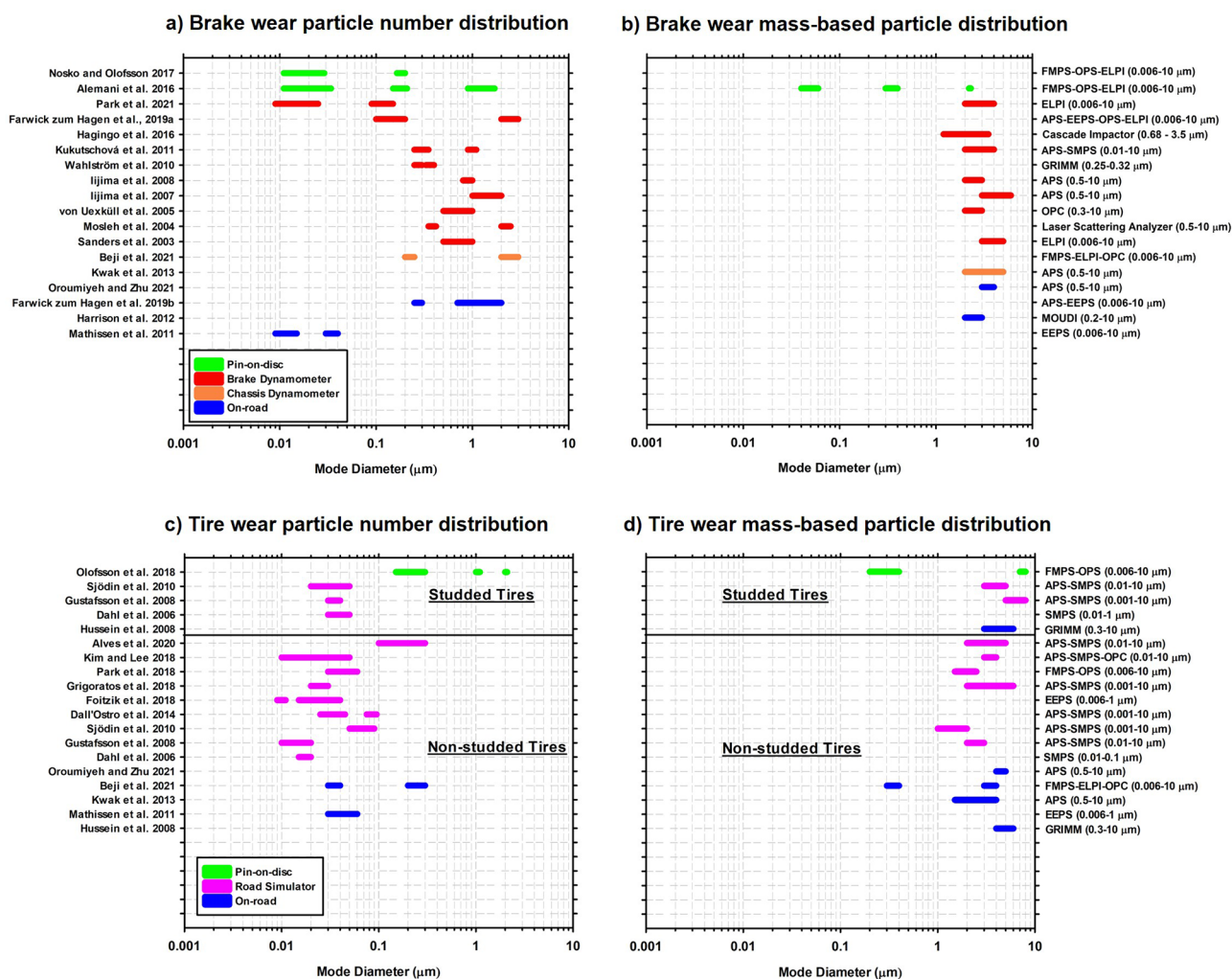


Figure 1. Overview of reported mode diameters of size distributions based on different measurement methods for (a) brake wear particle number distribution and (b) brake wear mass-based particle size distribution (c) tire wear particle number distribution, and (d) tire wear mass-based particle size distribution. Numbers in the parentheses show the detection size range of the measurement instruments: engine exhaust particle sizer (EEPS), aerodynamic particle sizer (APS), electrical low pressure impactor (ELPI), fast mobility particle sizer (FMPS), optical particle counter (OPC), laser scattering analyzer, optical particle sizer (OPS), scanning mobility particle sizer (SMPS).

penetrate deeper into the lung and possibly into the blood to impact other organs in the body.

In developed countries, tightening emission regulations for gasoline and diesel vehicles has mandated technological upgrades of combustion control and exhaust emission treatment systems. This has been effective in progressively driving down gaseous pollutants and PM from the exhaust of new vehicles. As a consequence, atmospheric emission inventories indicate that the proportion of NEE has increased,¹⁰ widely exceeding exhaust emissions.¹² In the UK, 2016 emissions data from the National Atmospheric Emissions Inventory (NAEI) showed that nonexhaust particles are the main source of primary PM (by mass) from road transport, for both the PM_{2.5} (60%) and PM₁₀ (73%) size fractions. A steady growth in this nonexhaust contribution is forecast, owing to phasing-out of older vehicles, increased electrification of road transport and the absence of legislation to limit/reduce nonexhaust particles. It is not surprising, therefore, that calls have been made for NEE from traffic to receive immediate recognition as an important source of ambient PM.¹²

Several reviews have been published on aspects of NEE,^{9,10,12,16,17} however, to our knowledge, previous articles have not incorporated health-related data, nor examined mechanisms driving NEE toxicity. Here we critically assimilate evidence on (a) the characterization of nonexhaust particles (Section 2), (b) their contributions to concentrations of, exposures to and health effects of ambient particulate matter (Sections 2–4), plus (c) the toxicological properties of this compositionally distinct source of particles (Section 5). We also address mitigation initiatives and identify critical research directions.

2. EMISSION CHARACTERISTICS AND QUANTIFICATION

NEE can be characterized in the laboratory at the subsystem level by testing individual material composites and different vehicle speeds/drive cycles on dynamometers.^{18–21} Alternatively, laboratory and on-road testing at the system level utilize real cars and chassis to assess specific variables under controlled test conditions,^{22–25} providing controlled and well-defined emission estimates. However, they are unable to characterize the entire vehicle fleet and variations in driving

styles, while climate and atmospheric processes may be relatively unrepresentative, causing discrepancies with real-world emissions.

Individual elemental tracers can be measured in the atmosphere and used to estimate NEE based on the material composition and/or scaling factors that account for real world emissions from the vehicle fleet under a range of meteorological conditions.^{26–28} However, this approach is potentially limited by wide variations in brake and tire material composites and is less able than laboratory studies to address specific vehicle emission attributes. Multivariate receptor modeling, such as Positive Matrix Factorization (PMF), can also be used to apportion chemical and/or size distribution data to the different source contributions. This has been used to identify NEE from atmospheric measurements,^{29–31} but studies (discussed in Section 3.1) have had limited success in separating individual NEE components and are likely to be location specific.

2.1. Non-Exhaust Emission Sources. 2.1.1. Brake Wear.

During a braking event, mechanical interaction between the brake pad and rotor produces brake wear particles (BWPs) of different sizes³² (Figure 1a,b). While mass-based BWP size distribution in the fine and ultrafine size ranges have been reported,³³ the majority of studies have reported a unimodal mass-based BWP size distribution with mode diameters in the range of 1–10 μm .^{24,34–38} Figure 1 illustrates greater variability, with a mode diameter number distribution from nanoscale to coarse size range. Reported number distributions are influenced by factors such as brake lining material^{19,39} and maintenance history.²³ Moreover, brake temperature can affect the BWP size distribution above a critical brake temperature ($140\text{ }^\circ\text{C} < T_{\text{crit}} < 240\text{ }^\circ\text{C}$) when ultrafine BWPs are generated.^{23,40–45} T_{crit} increases during multiple runs of laboratory analysis of BWPs, presumably due to differences in volatilization onset temperatures of brake wear organic materials.⁴² For instance, T_{crit} of 180 and 240 $^\circ\text{C}$ has been reported for brake pads with organic and inorganic binder contents, respectively.^{40,44}

Laboratory measurement studies have characterized the elemental composition of brake components.^{34,37,46–48} The metallic content of brake pads is dominated by Ba, Cu, Fe, Mn, Ti, and Zn.^{49,50} Other elements, including Al, Ca, Cd, Cr, K, Mo, Ni, Pb, Sb, Si, Sn, and Zr, have also been reported.^{37,51–54} In addition, while rare-earth elements were previously reported to be predominantly associated with mineral dust,^{55,56} a group including Gd, Ho, Lu, Pr, and Tb have been recently found in brake pads.⁵⁷ The total carbonaceous fraction of BWPs can be highly variable (5–76%) depending on the brake pad material, braking velocity, and brake temperature.^{21,58,59} Approximately 150 organic compounds have been identified including *n*-alkanes, *n*-alkenes, *n*-alkanols, glycerol compounds, phenolic compounds, and polycyclic aromatic hydrocarbons (PAHs).²¹ Concentrations are higher during light- than heavy-braking events, indicative of thermal degradation.^{21,60} For brake discs, Fe has been reported to be the most dominant element.⁵⁰

Elements most frequently used and deemed to be the most specific as brake wear tracers for source apportionment of fine and coarse particles are Ba, Cu, Sb, and Sn.^{34,61–64} Although others (e.g., As, Cr, Fe, Mn, Mo, Sr, Ti, Zn, and Zr have also been recommended^{11,65–67}), many have been associated with other emission sources. For instance, Zn, Fe, Ti, Mn, and Sr can be found in tire wear, industrial emissions, and mineral dust.^{11,56,68–70}

2.1.2. Tire Wear. Variables influencing tire wear include tire characteristics (e.g., composition, construction, studs), road surface characteristics and vehicle operation/characteristics (e.g., speed, cornering, weight, power).⁷¹ Composition is also influenced by heat generated and the incorporation of other particles, most notably road surface material.^{72–74} Kreider et al. developed a nomenclature describing whether particles were generated from the original tread (TP), laboratory-generated tire wear particles (WP) or on-road collected particles (RP).⁷³ The abbreviation TRWP (tire- and road wear particles) also encompasses the combination of these sources from a range of urban environments.⁷⁵ A broad size range (10 nm to 10 μm) of laboratory generated TRWP has been measured (Figure 1c,d),^{76–82} with a unimodal mass distribution in the 1–7 μm range and number size distribution of 10–200 nm. On-road TRWP sampling and measurements on instrumented cars have adopted various approaches.^{83–85} Studies based on the TRAKER method report that particles sampled behind the front wheel of a car have a size distribution peaking at about 2–3 μm , with a high content of crustal elements, indicating a high contribution from road wear and dust.^{38,86} Beji et al, using electrical low pressure impactor devices, measured a bimodal size distribution (<0.03 μm and 0.05–0.30 μm by number; 0.1–0.6 μm and 1.0–15 μm by mass) when sampling from the front tire.²⁵ Studded tires, used in cold climates to improve grip, generate ultrafine particles (<100 nm) (Figure 1c,d), that is speculated to originate from evaporation and subsequent condensation of softening oils in the rubber mix.⁸⁷

Only 1% of tire wear is estimated to be released into the PM_{10} fraction,⁷³ with varying contribution to atmospheric PM_{10} : <1% using tire tread polymers as tracers,⁷⁵ 0.1–3.9%⁸¹ and 1.0–7.5%⁸⁸ using source receptor techniques, and 3–4% using simultaneous measurements of mass behind the front wheel and in the surrounding atmosphere.³⁸ Few estimates of the contribution of tire wear particles to airborne concentrations exist, but available data indicate for the $\text{PM}_{2.5}$ fraction: 0.1–0.68% of mass in London, Tokyo and Los Angeles using tire tread polymers⁷⁴ and 4–7% using the mass based technique.³⁸

Tires typically constitute rubbers/elastomers, fillers, processing oils, additives, reinforcements, and vulcanization agents, which vary depending on end use.⁸⁹ A truck tire contains 80% natural rubber, whereas passenger car tires contain only 15%.⁹⁰ Some of these can be used as markers for tire wear in the environment. Examples include components used in the vulcanization process, such as 2-(4-morpholinyl)-benzothiazole⁹¹ and Zn,⁹² or those originating from thermal decomposition of tire tread polymers⁹³ such as styrene, isoprene, dipentene, butadiene, vinylcyclohexene, and benzothiazole. The latter is used in the ISO standard for determination of TRWP.⁹⁴ Compared to TP, TRWP are enriched in metals from brake linings and pavement materials and contain a lower concentration of polymers.^{72,73} Zn is a notable exception, which is enriched in TP relative to TRWP, despite being emitted by other sources such as industrial processes⁷⁰ and brake pads.⁵⁰

2.1.3. Road Wear. Road surfaces are predominantly asphalt mixtures of ballast rock aggregates bound together with a bituminous binder, with several additives to improve durability such as fibers, resins, filler mineral powder, and polymers. In countries where studded tires are used, pavements have a high content of coarse wear-resistant rock aggregates in stone mastic asphalts. Smaller, less durable rock aggregates in less dense

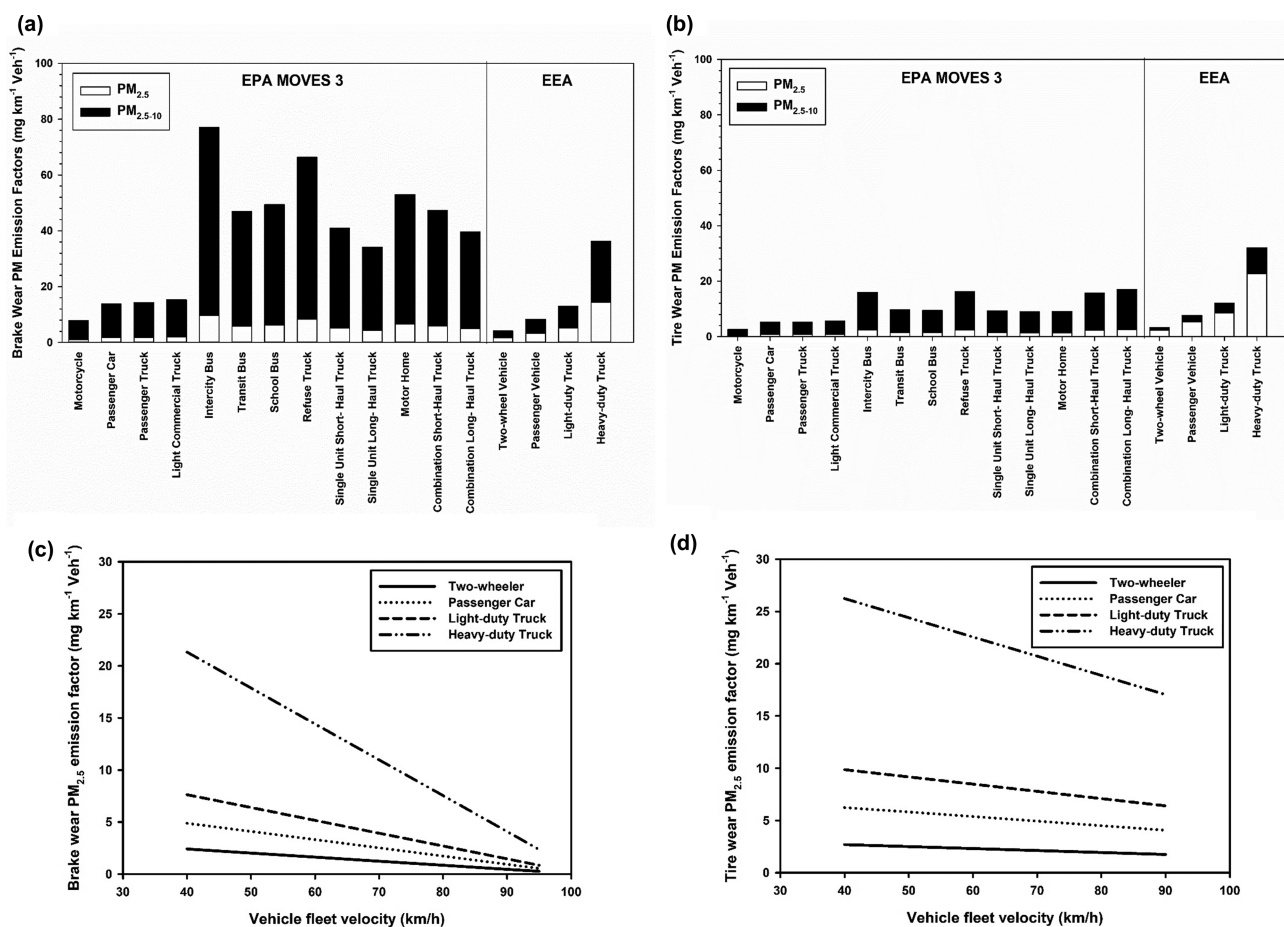


Figure 2. Effect of fleet velocity and weight on brake and tire wear PM emission factors (EFs) estimated by United States Environmental Protection Agency (USEPA) and European Monitoring and Evaluation Programme (EMEP)/European Environment Agency (EEA). Effect of weight on (a) brake and (b) tire wear PM EFs. Effect of vehicle velocity on PM_{2.5} EFs of (c) brake and (d) tire wear particles. EMEP/EEA brake wear PM EFs estimated for vehicle speed of 60 km/h EMEP/EEA heavy-duty truck brake wear PM EFs estimated for a half full truck with four axles.

constructions can be used where studded tires are not common. The geographical availability of high-quality rocks also determines what is economically and environmentally feasible to use.

As vehicles tires move over the road surface, the interaction generates wear of both materials. Road wear particles (RWPs) are dominated by aggregate rock minerals that normally cover over 90% of the surface. Size distribution starts at $\sim 0.2 \mu\text{m}$ but particles are mainly in the coarse mode. Tests using road simulators, with and without studded tires, report a mean mass size distribution peaking at 6–8 μm ^{95,96} and a bimodal appearance with mass peaks at 2–3 and 7–8 μm .⁹⁷

Since most of wear dust originates from local crustal rocks, specific tracers are difficult to define, except in controlled studies where rocks with traceable composition are used.⁹⁸ Abundant elements include Si, Al, Ca, K, Fe, and Ti. Size separated analyses suggest that these mineral related elements are abundant above $\sim 1 \mu\text{m}$. Elements such as S and Cl, attributed to the binder matrix (bitumen), are more abundant below 1 μm ,⁹⁹ although these could also be related to tire wear. The bitumen of asphalt pavements contains a mixture of high molecular weight organic compounds. Approximately 5% of total suspended particles (TSP) in roadside samples in Denmark were bitumen particles.¹⁰⁰

2.1.4. Resuspension. Road dust consists of particles present on the road surface, generated by traffic or transported and deposited from near or long-range sources. Brake wear particles, RWP and TRWP originating from road and traffic, are major components of road dust,^{101,102} but winter traction sanding, building sites, wind-blown dust from bare soils, and dust dragged in by traffic from connecting unbound roads can be strong local sources. Source heterogeneity is reflected in the chemical composition (a mix of minerals, metals, and organic compounds) often occurring as aggregates with a high temporal and spatial variability. The RD10 (road dust smaller than 10 μm) fraction sampled in Oporto, Portugal, contained 11% organic carbon (OC) and 5% elemental carbon (EC).¹⁰³ Metal oxides accounted for 30% and 73% in samples from asphalt and cobbled roads, respectively.¹⁰³ The fraction contained hundreds of organic compounds of which plasticizers were the most abundant. Traffic related elements, such as Fe, Zn, Cu, Ba, and Sb are normally enriched in road dust in urban areas.¹⁰⁴

The resuspension (or emission) of road dust is a complex process, dependent on road dust load,¹⁰⁵ road surface macro texture,¹⁰⁶ and humidity,¹⁰⁷ traffic density, composition, and vehicle speed. Road dust load is a function of source strengths and the surfaces' ability to contain dust under prevailing meteorological and traffic conditions and varies highly in time

and space. A high macro texture can store more road dust but will make it less available for resuspension and thereby reduce emissions.¹⁰⁸ Resuspension of dust is negligible when the road surface is moist and after rain events, and mobile dust load recovers faster in warm, dry climates compared to more humid ones.¹⁰⁹ Water can also react with road dust, causing a cemented dust load in the texture that needs high suspension forces to be resuspended.

Resuspension of road dust is a strong PM source, especially in dry regions and where studded tires and winter maintenance (e.g., use of traction sand) are used. In Delhi, India, resuspension is estimated to account for 79% of PM¹¹⁰ compared with ~38% of the traffic increment of coarse particles above the urban background, at a curbside site on a congested central London street.⁹² In Nordic regions, road dust accumulates during winter when sources are strong and the road surface is humid or frozen, causing a typical resuspension PM₁₀ peak in early spring.^{105,111,112} Cross-street sampling demonstrates high variability with low loads in wheel tracks and high loads between/on curbs, implying differing resuspension potential depending on whether vehicles keep to the wheel tracks.¹⁰⁵ PMF modeling estimates the composition of RD10 in Paris streets to be equal percentages of road wear, brake wear, and carbonaceous dust.⁵⁵

2.2. Emission Factors. Nonexhaust emission factors (EFs) predict at wider geographical and temporal resolutions than can be achieved by measurements alone. They are typically presented as mg·km⁻¹·veh⁻¹ for different vehicle classes for the purpose of national atmospheric emission inventory reporting and can cover a range of driving styles. While distance-based EFs can account for speed and meteorological conditions (e.g., by applying correction factors), they are not dynamic and therefore constrained in providing very localized emission predictions, such as in areas with high levels of acceleration/braking.

2.2.1. Brake and Tire Wear. Variability in estimated brake and tire wear PM EFs derived from different sampling approaches has called for updated emission inventories, taking into account various factors (e.g., braking activity, average deceleration rate, vehicle weight) based on results of multiple studies. The UK NAEI reported an average brake and tire wear PM_{2.5} EF of 3 and 5 mg·km⁻¹·veh⁻¹, respectively, and brake and tire wear PM₁₀ EFs to be 7 mg·km⁻¹·veh⁻¹ (NAEI, 2018). Emission factors provided by the United States Environmental Protection Agency (USEPA), under the Motor Vehicle Emission Simulator (MOVES) program (Figure 2),¹¹³ for brake wear, PM_{2.5} and PM₁₀ range from 1.0 to 9.6 and 7.8–77.0 mg·km⁻¹·veh⁻¹, respectively, and for tire wear PM_{2.5} and PM₁₀ range from 0.4 to 2.6 and 2.7–17.1 mg·km⁻¹·veh⁻¹, respectively. Using the MOVES model, intercity bus and refuse trucks have the highest brake wear PM_{2.5} and PM₁₀ EFs (Figure 2), exceeding those heavier vehicles (e.g., combination trucks), demonstrating the influence of frequent vehicle braking activity.

The European Monitoring and Evaluation Programme (EMEP)/European Environment Agency (EEA) proposed the most comprehensive method for estimating brake and tire wear, PM_{2.5} and PM₁₀ EFs, based on TSP concentration and incorporating vehicle weight and velocity.¹¹⁴ For a similar vehicle category, EMEP/EEA reported higher brake wear PM_{2.5} EFs, but lower PM₁₀ EFs (Figure 2) compared to the USEPA data, and higher tire wear PM_{2.5} and PM₁₀ EFs than USEPA for any given vehicle class. This is likely due to the

MOVES model estimating EFs from a limited number of older studies, assuming brake wear PM₁₀ EFs to be ~8 times higher than PM_{2.5},^{19,113} and using a tire wear PM₁₀/PM_{2.5} ratio of 6.7 (recent studies have shown a tire wear PM₁₀/PM_{2.5} ratio of 2.0–2.5).^{77,113} Incorporating the impact of vehicle velocity (Figure 2b,c), EMEP/EEA proposed negative linear relationships between the vehicle fleet velocity and brake and tire wear EFs,¹¹⁴ which have value for identifying brake and tire wear hotspots in urban environments.

2.2.2. Road Wear. Among NEE, road wear is the least investigated, partly because direct emissions are difficult to separate from resuspended road dust. Using EMEP/EEA inventories and models, PM₁₀ road wear direct EFs for passenger cars range from 5–10 mg·km⁻¹·veh⁻¹.^{114,115} A summary of five studies from road simulators, on-road measurements, and emission inventories, generated median EFs for light duty vehicles (LDV) of 7.75 mg·km⁻¹·veh⁻¹ and for heavy duty vehicles (HDV) 33.5 mg·km⁻¹·veh⁻¹.⁹ In countries where studded tires are used the road wear EF is harder to separate from resuspension, but has been estimated as 200–400 mg·km⁻¹·veh⁻¹.⁹⁹

2.2.3. Resuspension. PM₁₀ EFs for resuspension on paved roads have been estimated using derivations from dust load, mobile measurements and vertical profile methods. Factors range from a few to ~1000 mg·km⁻¹·veh⁻¹ for LDV¹⁰² and 150–8000 mg·km⁻¹·veh⁻¹ for HDV¹¹⁶ (using USEPA method based on silt loading and vertical profile of dust loading). PM₁₀ resuspension was between 13 and 32 mg·km⁻¹·veh⁻¹ for 10 sites in Milan, Italy.¹¹⁷ A CO₂ dilution approach determined average vehicle fleet resuspension EFs for dry (4–11 mg·km⁻¹·veh⁻¹) and moist/wet road surface conditions (2–7 mg·km⁻¹·veh⁻¹) on a congested London road, UK.²⁶ The high variability is a result of geographical differences and possibly sampling, measuring and emission factor estimation methods. Modeling resuspension approaches include simple models using silt load in combination with mean traffic vehicle weight as the main influencing parameters,¹¹⁸ whereas more recent approaches take into account other factors (e.g., surface humidity and texture properties).¹¹⁹

3. EXPOSURE ASSESSMENT OF NON-EXHAUST PARTICULATE MATTER

Exposure assessment of nonexhaust PM and sources is an emerging discipline. Estimation of airborne concentrations has used elemental tracers,¹²⁰ but these are subject to interferences as no element is a wholly specific source tracer. Organic compounds have been used as tracers of tire wear,¹²¹ and pyrolysis/gas chromatography to quantify rubber.¹²² Element combinations may also be used in receptor modeling methods to identify factors consistent with nonexhaust sources.¹²³ However, high levels of uncertainty surround all measurement methods. Common approaches for short- and long-term epidemiological studies include collection of monitoring data (from central stations, in-cabin vehicles, personal monitors), and use of high-resolution model predictions (from dispersion/chemical transport and land use regression (LUR)). Compared to the majority of studies on single air pollutant exposure (e.g., PM_{2.5} and gaseous pollutants), assessing nonexhaust exposure is more complex, requiring precise apportionment of PM mixtures for nonexhaust sources and accurate characterization of exposure for individual participants.

3.1. Methods for Assessing Short-Term Exposure to Nonexhaust Sources and Constituents. Short-term studies of associations between hospital admissions, emergency department visits and mortality, and exposure to daily PM constituents and sources have typically used data (e.g. trace metals, EC and OC) from one central monitoring station per city to conduct source apportionment analyses.^{124–132} Strengths include long-term continuous data (usually >4 years) of high quality, allowing robust source classifications and estimations generated by a source apportionment model. The PMF model has been the most used given its flexibility to generate source contribution estimates without relying on a priori knowledge of source chemical profiles. Methods such as chemical mass balance (CMB),^{123,130} multilinear engine 2,¹³² and a hybrid model with CMB and non-negative factor analysis,¹³¹ have also been applied.

Many exposure studies have captured road dust as an indicator of NEE sources and linked it to epidemiological analyses. However, further separation of individual emissions has not been attainable from the PMF model, with only one exception, a study in Barcelona where the brake wear factor (Fe–Cu–Sb-rich particles) was disentangled from the mineral and road dust combinations in PM₁₀ and PM_{2.5}.¹³² One possible reason behind the difficulty in separating NEE source factors is that, because of the health relevance and data availability, many studies focus on exposure to constituents of PM_{2.5} rather than those in coarse particles (PM_{2.5–10}), which form a significant part of NEE PM mass. Furthermore, depending on the representation of city monitoring sites, studies may not sufficiently capture near-roadway traffic-related sources, but rather indicate urban background mixtures. Lastly, many sources of NEE are spatially and temporally correlated.¹²³

Directly measuring PM exposure in vehicles, or use of personal monitors may better capture specific NEE sources for epidemiological studies. A U.S. study measuring in-cabin exposure of patrol police officers to PM_{2.5} constituents identified two NEE sources indicating automotive steel wear (Ti–Cr–Fe particles) and speed-changing traffic (dominated by aldehydes–S–Cu).¹³³ In the Atlanta Commuters Exposure (ACE) study, in-vehicle exposure to NEE were characterized by Ba, Fe, Mn, and S suggesting a strong contribution from brake wear.¹²³ In a study in Hong Kong using personal exposure data and a PMF model, dust-related pollution was separated into crustal/road dust and a nonexhaust traffic-related fraction.¹³⁴ It should be noted that, unlike ambient monitoring data, in-vehicle and personal exposure data are not geographically fixed. Therefore, these analyses rely on the assumption that the sources are the same across the driving/walking routes, which may complicate the interpretation of the source identification results.

3.2. Methods for Assessing Long-Term Exposure to Nonexhaust Sources and Constituents. Approaches taken to estimate long-term exposure to NEE sources and constituents include dispersion/chemistry-transport or LUR models. These are typically useful in characterizing intraurban contrasts at a small scale, such as the dispersion pattern of local traffic air pollution in large population studies with high spatial resolution.

Two popular dispersion models with capability to predict NEE include the UK KCLurban model developed for the Greater London area¹³⁵ and a Swedish dispersion model primarily used in Stockholm, Umea and Gothenburg.¹³⁶ These

systems incorporate a detailed local emission inventory allowing apportionment of different local sources, such as the differentiation of exhaust and NEE from road traffic. Since the contribution of NEE in PM_{2.5} versus PM₁₀ was empirically determined using field monitoring campaign data (in the Swedish model, 20–30% of PM₁₀ was PM_{2.5}), the uncertainty of the EFs is potentially larger for nonexhaust PM_{2.5} than total fine particles. To evaluate this uncertainty for PM_{2.5} and PM₁₀ brake wear, tire wear, and resuspension, source apportionment estimates have been compared against observations from urban monitoring sites or field campaigns in a street canyon.^{27,112,119} Both models provide annual average estimations for non-exhaust PM_{2.5} and PM₁₀ for many years on a regular 20 × 20 m (UK model) or 50 × 50 m (Sweden model) grid. This distinguishes exposures for a sizable population within a city, although several limitations remain. First, exhaust and nonexhaust PM exposures are highly correlated in their spatial dispersion due to their common generation presenting difficulties for discerning independent health effects of the individual sources. Second, model uncertainty that implies spatial agreement of annual mean prediction and observation data in long-term exposure studies presents challenges due to the small number of validation sites. Validation studies that incorporate additional monitoring data from field campaigns are therefore needed. Lastly, the UK and Sweden models are unique in terms of their detailed local emission inventory to apportioning exhaust and nonexhaust sources, and generalization to other areas with imperfect emission inventories is challenging.

For several cohort studies in California,, trace metals in PM_{2.5} (including Fe and Zn as indicators of NEE) have been estimated by a chemistry-transport model.^{137,138} Compared to dispersion models in Europe, these provide daily predictions over many years that are useful for both short- and long-term studies.¹³⁹ While the model has performed moderately well for Fe and Zn across individual sites across cities, the coarse spatial resolution (4 × 4 km) prohibits investigation of fine-scale variations of traffic-related exposure.

LUR models are commonly used to assess long-term air pollution exposure,¹⁴⁰ predicting spatial variations of trace metals in PM as markers of different sources in North America,^{141–148} Europe,^{149,150} Australia,¹⁵¹ and East Asia.^{152,153} These are empirically based on the relationship between measured air pollution and a number of geographical predictor variables, and estimate exposures at individual residential locations. Due to limited numbers of regular monitors for PM composition in a city, dedicated filter-based monitoring networks consisting of spatially dense (20–150 sites per city), but temporally sparse (7–14 sampling days in 2–3 seasons of a year) sites are often established to capture traffic-related PM sources and spatial dispersion. In this way LUR models are typically developed for annual or seasonal average predictions. Brake/tire wear PM composition markers have been determined either based on a priori knowledge from former studies (e.g., Cu, Fe, Zn in the ESCAPE (European Cohort Study for Air Pollution Effects)¹⁵⁰ and MESA Air (Multi-Ethnic Study of Atherosclerosis and Air Pollution)¹⁴⁸), or associations between trace metals in PM and traffic variables within small distance buffers (e.g., Ti, Cu, Fe in a New York city study).¹⁴¹ Overall, studies have shown that LUR models perform moderately well in explaining within-city variability of nonexhaust related trace metals (e.g., Cu, Fe, Ti, Ba, Zn) in PM₁ (particulate matter less than 1 μm in diameter) PM_{2.5} and

PM₁₀ although less so than models for air pollutants that represent exhaust emissions.^{154,155} Potential challenges include imperfect study design, predictor variables, and modeling algorithms. For example, traffic-related predictor variables used in LUR models (e.g., road length, traffic volume, and near-road distance) cannot effectively disentangle exhaust and non-exhaust PM composition, preventing attempts to attribute health effects to separate sources. Predictor variables that reflect activities to produce/reduce brake and tire abrasion (e.g., road intersections, traffic speed) and road dust resuspension (e.g., use of studded tires, street cleaning, maintenance activities) are essential to improve nonexhaust LUR models. Recent studies have shown that there may also be value in incorporating machine-learning approaches to further improve model performance, since the dispersion pattern of NEE could be more local and highly nonlinear than exhaust pollutants.^{149,156} It is noteworthy that trace metals such as Cu, Fe, and Zn may not exclusively represent vehicle sources, with LUR models in Europe and Canada reflecting additional emissions released by metallurgical industries.^{147,149} Two U.S. studies have taken a further step by combining a source apportionment model with a LUR model to address the spatial variations of PM mixtures instead of focusing on individual constituents.^{144,145} Although both studies identified a brake wear source across the sites, the predictor variables were too general to explain source variations to a sufficient extent.

4. EPIDEMIOLOGICAL STUDIES

Although it is postulated that chemical composition may better explain observed PM-related health effects than mass alone,¹⁵⁷ epidemiological studies of specific chemical components of PM, including constituents of NEE, remain scarce, in part due to limitations in exposure assessment. Studies have not yielded consistent results, with chemical tracers for combustion derived PM (EC, black carbon, Cr, V, Ni) often being more predictive^{120,158} and as a consequence, calls have been made for further studies.^{159,160} Studies examining PM composition have focused on a large set of species from ions (SO₄²⁻, NO₃⁻) and carbonaceous components (EC, OC) to metals (e.g., Ni, V, Cu, Zn, Fe). By and large when examining health effects, individual components are considered as tracers for sources. For example, and as discussed previously, in terms of NEE, Cu is typically used as a tracer for brake wear, Zn for tire wear (although there are industrial sources),¹⁴⁹ and Fe for resuspended road dust.¹⁶¹ Epidemiological studies using source profiles obtained from dimension reduction methods including PMF often do not resolve a specific NEE source, but rather a traffic (primarily EC) or metal profile.¹⁶²

4.1. Short-Term Studies. Time series and case-crossover models are typically used in assessing short-term associations with mortality or morbidity as a function of exposure adjusted for time and time-varying meteorological factors (e.g., temperature, humidity). Effect estimates are reported as odds ratios (OR) or percent increases per 10 μg/m³, or per interquartile range (IQR) change in PM or PM species. NEE components are examined directly or accounted for as a proportion of the mass that modifies the PM association (e.g., 0.75% excess risk (ER) in the PM_{2.5} estimate when the proportion of mass is higher in Zn). When components are examined directly, the models are generally adjusted for PM mass, but not other closely associated pollutants and constituents.

A systematic review synthesizing 32 short-term studies (31 time series, 1 case-crossover) found statistically significant pooled ER of cardiovascular-related mortality (ER = 0.49%, 95% CI 0.03–0.96%) per IQR increase in PM_{2.5} Zn.¹⁵⁹ It also reported significant heterogeneity in cardiovascular morbidity with PM_{2.5} Fe and Zn (Cochran's *I*² = 52% and 62%, respectively). A study of Canadian hospital admissions found increased cardiovascular events in men associated with Cu, Fe, and Zn exposures when increased sulfur concentrations were also present (e.g., Cu OR = 1.08, 95% CI 1.05–1.11 per 10 μg/m³).¹⁶³

The few epidemiological studies to use receptor modeling to identify a NEE component have not separately identified various sources, but report a single component, often referred to as “road dust”, but probably also containing other NEE sources. In terms of source contributions, both total and cardiovascular mortality were associated with 2-day lag exposure to road dust (loading on Cu–Fe–Zn–S) in a study of deaths in Barcelona, Spain.¹⁶⁴ Rich et al. found IQR increases in road dust (containing Cu, Fe, Zn) on the same day were associated with significantly increased ischemic heart disease hospitalizations (ER = 0.6%, 95% CI 0.1–1.1%).¹²⁴ Although not statistically significant, most NEE-associated ER of congestive heart failure, ischemic heart disease and myocardial infarction on the same day and previous 4 and 7 days were positive. A panel study of an identified NEE source representing “speed-changing traffic and with engine emissions and brake wear” (dominated by aldehydes–S–Cu) showed a strong association with heart rate variability in nine healthy, nonsmoking male highway patrol troopers.¹⁵³

The aforementioned systematic review did not find any NEE markers to have significant pooled associations with respiratory mortality or morbidity.¹⁵⁹ However, in a recent study of 100 patients with chronic obstructive pulmonary disease in Shanghai, China, greater levels of Cu and Fe were associated with a reduction in forced expiratory volume in 1s (FEV₁) and forced vital capacity (FVC), and Cu was also linked to reduced peak expiratory flow.¹⁶⁵ Similarly, a source factor representing NEE (loading on Ba–Fe–Mn–S) was associated with decreased FEV₁ (–0.84% 95% CI –2.27–0.58) and increased airway inflammation in commuters in Atlanta, GA.¹²³ The ACE study showed Cu to be significantly associated with increased exhaled nitric oxide and decreased FVC in 60 young adults.¹⁶⁶

4.2. Long-Term (Chronic) Studies. Long-term (often cohort) studies are typically conducted using Cox proportional hazards models. Effect estimates are reported as hazard or risk ratios (HR/RR), or a change in risk associated with 10 μg/m³ or IQR increase in PM or PM species. Yang et al. synthesized 11 cohort studies and found significant pooled associations of nonaccidental mortality per IQR increase in PM_{2.5} Zn (ER = 9.4%, 95% CI 6.33%–12.56%), as well as cardiovascular mortality per IQR increase in PM_{2.5} Fe and Zn.¹⁵⁹ Significant heterogeneity in PM_{2.5} Fe and Cu and cardiovascular mortality was also found (*I*² = 95% and 90%, respectively). The ESCAPE studies found an increased relative risk of cardiovascular mortality with PM_{2.5} Cu (RR = 1.005, 95% CI 1.001–1.009) and respiratory mortality with PM₁₀ Zn (RR = 1.136, 95% CI 1.010–1.277) in England, UK.¹⁶¹ In a pooled analysis of the 8 ESCAPE cohorts, robust associations were found between all-cause mortality and linear regression modeled PM_{2.5} Cu, Fe, and Zn in single pollutant models, and when adjusted for PM_{2.5} mass.¹⁶⁷ Cardiovascular and lung-cancer mortality showed

similar RRs for these NEE components, but they did not remain statistically significant when adjusted for $PM_{2.5}$ mass. In a population-based Canadian cohort, long-term exposure to $PM_{2.5}$ Fe and Cu were consistently associated with increased cardiovascular mortality (Fe HR = 1.014, 95% CI 1.008–1.019; Cu HR = 1.006, 95% CI 1.001–1.012) and congestive heart failure (Fe HR = 1.031, 95% CI 1.024–1.038; Cu HR = 1.015, 95% CI 1.009–1.022).¹⁶⁸

Fuertes et al. found that across seven European birth cohort studies, PM_{10} Zn was the only element independently associated with a higher risk of early life pneumonia (OR = 1.47, 95% CI 0.99–2.18).¹⁶⁹ In a Dutch birth cohort, asthma symptoms were positively associated with PM_{10} Cu (1.06, 95% CI 1.00–1.12), as was decreased FEV_1 (–2.3%, 95% CI –4.3–0.3%).¹⁷⁰ PM_{10} Cu and Fe were associated with increased allergic sensitization, and $PM_{2.5}$ Cu and Fe were associated with decreased FEV_1 .

A meta-analysis of 32 studies found a significant decrease in birth weight per IQR increase in $PM_{2.5}$ Zn (pooled effect –7.5 g, 95% CI –10.0 g to 5.0 g).¹⁷¹ A UK study found elevated risk in preterm birth associated with nonexhaust $PM_{2.5}$, modeled as brake/tire wear and resuspension of road dust (OR = 1.03, 95% CI 1.01–1.05).¹⁷² In multipollutant models, elevated $PM_{2.5}$ Zn was associated with higher mean diffusivity (a marker of neurodevelopment linked to psychiatric and neurological disorders) in Danish children.¹⁷³ Another study reported an increased OR of autistic disorder in children by age 6 years when mothers were exposed to elevated concentrations of Cu during pregnancy.¹⁷⁴

In summary, short- and long-term studies of $PM_{2.5}$ and PM_{10} elemental composition have been conducted, some focusing specifically on certain NEE tracer elements or source factors. There is consistency in the findings for PM Zn and road dust source factors being associated with acute and chronic cardiovascular outcomes, as well as birth outcomes. Several components of NEE showed significant associations with children's respiratory health and neurodevelopment.

5. TOXICOLOGY/MECHANISTIC DATA

The physicochemical properties of NEE indicate considerable potential to cause adverse biological responses if inhaled, either through activation of resident airway inflammatory cells, or via interactions with the underlying epithelium. However, compared with the extensive literature on combustion particle toxicity, relatively few studies have directly addressed the hazards associated with NEE exposure. Those that have, mostly focus on the capacity of NEE-derived PM to induce oxidative stress as a trigger for acute inflammation and cellular injury, with a focus on metals and organic species derived from mechanical wear.¹⁷⁵ These species can catalyze the formation of ROS, either directly, through catalysis of the Fenton and Haber-Weiss reaction (Fe, Cu, Mn, etc.)¹⁷⁶ or indirectly (in the case of Zn) by interfering with intracellular signaling through inhibition of phosphatases,¹⁷⁷ through the generation of reactive electrophiles and ROS through the induction of xenobiotic pathways (organic components),¹⁷⁸ or disrupting mitochondrial function.¹⁷⁹

5.1. NEE in Roadside PM. The largest body of work that indirectly examined NEE toxicity compared roadside (enriched with NEE source contributions) versus urban background/rural PM, relative to source-specific reference PM standards (typically primary diesel exhaust particles (DEP)). Researchers have generally attempted to dissect out contributions of

individual components of NEE through correlating the range of toxicological end points to imperfect chemical tracers (usually metals/metalloids: Zn for TWP; Cu, Ba, Sb, and Sn for BWP; Fe, Cu, and Si for RWP; versus Ni, Cr, and V for oil combustion). As many of these components are highly correlated within the panels of PM tested, this method has inherent limitations. However, several studies augmented this observational approach through inhibition of component-specific biological pathways (metal handling,^{180,181} aryl hydrocarbon receptor mediated xenobiotic metabolism,¹⁸⁰ toll-receptor mediated responses to PM associated biological components¹⁸² or PM fractionation¹⁸³).

Several in vivo animal studies examined the NEE contribution to RWP toxicity. Gerlofs-Nijland et al. comparing the effects of coarse ($PM_{2.5-10}$) versus fine ($PM_{0.18-2.5}$ accumulation mode) RWP collected at various European sites, in rats following pulmonary instillation.¹⁸⁴ The results were complex but did show that PM from traffic heavy sites induced pulmonary inflammation and raised circulating concentrations of blood fibrinogen (an acute phase response protein involved in coagulation). These effects were correlated with Cu, B, and Zn concentrations (reflective of mixed NEE sources), but not PAH content. In many cases, effects were only seen at the highest exposure levels (~2.5 mg/rat) and the authors noted the actions of certain markers could have arisen from sources such as wood-smoke. Happonen and colleagues (2010) investigated different sized urban PM samples ($PM_{10-2.5}$, $PM_{2.5-1}$, $PM_{1-0.2}$, $PM_{0.2}$) from Helsinki, Finland,¹⁸⁵ collected at different seasons to capture spring-time increases in $PM_{10-2.5}$, caused by resuspension of winter road dust. Selected elements were used as indicators of NEE (road dust: Ca, Fe, Al, Mn; tire and brake: Cu, Zn, Fe). Instillation in mice (~0.25 mg/mouse) resulted in pulmonary inflammation, with NEE-enriched samples inducing the most pronounced response, especially with coarse mode PM containing elevated endotoxin (a bacterial product). These samples also exhibited relatively high EC/OC ratios (indicative of combustion-derived PM), making it challenging to separate the effects of exhaust and NEE. Nonetheless, the authors concluded traffic-derived PM from mixed NEE sources was the key driver of the inflammatory responses. Kreider and colleagues (2012) carried out a subacute inhalation study of mixed TWP and RWP in rats.¹⁸⁶ Lower doses were tested (10–100 $\mu\text{g}/\text{m}^3$, 6 h/day for 28 days), with no evidence of enhanced mortality or gross pathology effects. Occasionally foci of lung pathology were observed with high exposures (100 $\mu\text{g}/\text{m}^3$), but were unaccompanied by markers of lung or systemic inflammation, oxidative stress, or thrombosis. The authors concluded a no-observed-adverse-effect level of 112 $\mu\text{g}/\text{m}^3$ - well above environmental levels, for this animal model (the potential human implications are discussed in Kreider et al.¹⁸⁷). The study also contained unpublished instillation data in the discussion that compared different types of PM. PM from tire-brake wear did not significantly increase pulmonary inflammation, whereas DEP or crystalline silica did. The authors therefore suggested that, despite TWP and BWP presenting a significant fraction of PM mass, they did not contribute significantly to ambient PM toxicity.

5.2. Toxicity of Brake Wear. Increasingly, in vitro and in vivo evidence suggests that metals within BWP (specifically those capable of catalyzing ROS production), are important contributors of toxicity. This was first addressed by Gasser et al. using freshly generated BWP and an alveolar epithelial cell

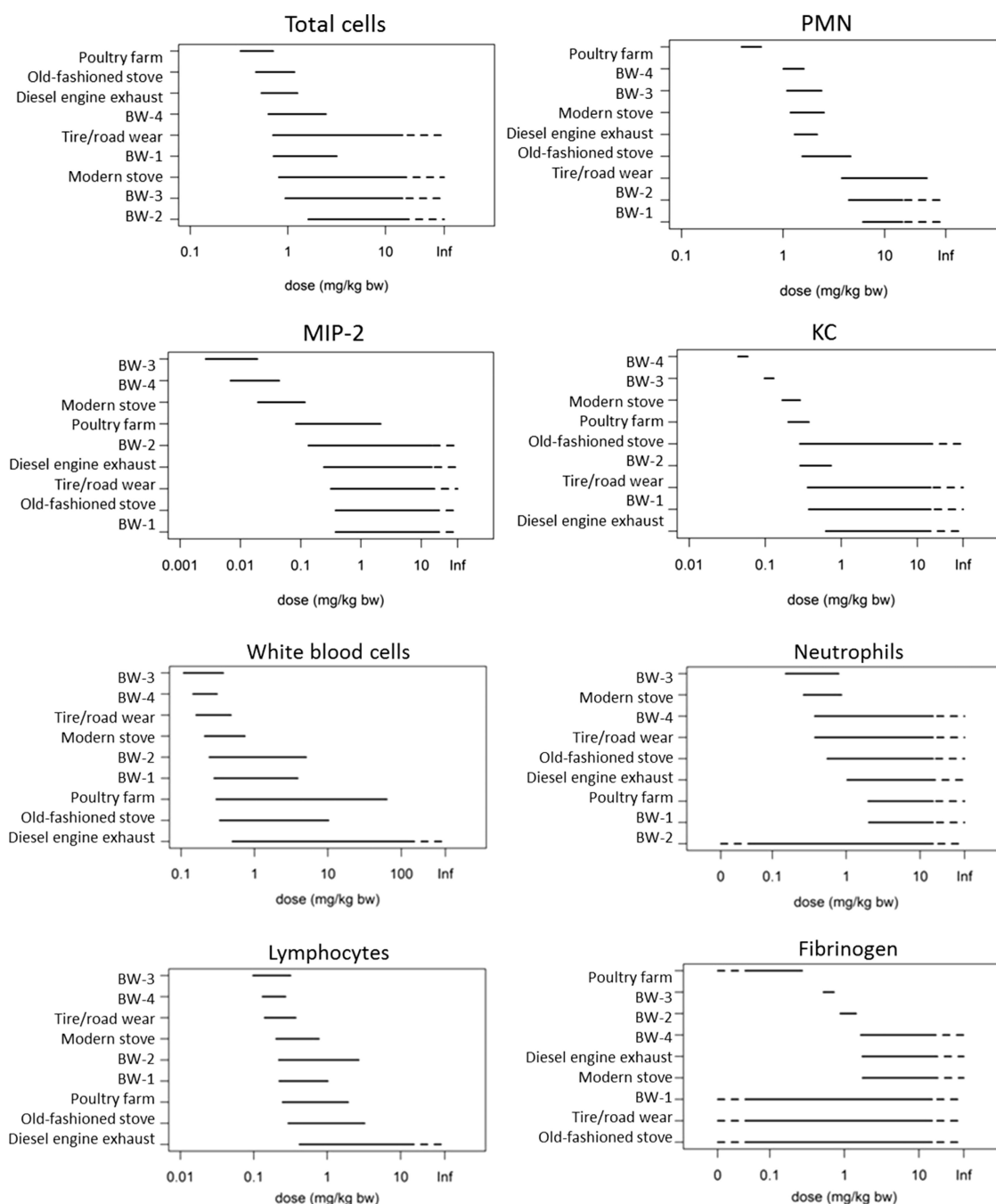


Figure 3. Relative toxicity of BWP compared to particulates from other sources following inhalation in mice. The further the line is to the left, the lower the dose at which it exerts an effect (i.e., the greater the toxicity) for that given parameter. In several assays, BWP had a greater effect than tire wear, DEP, and other PM. Equally, the relative toxicity varied considerably between the parameter being studied. Upper four panels are measures of pulmonary inflammation, lower four panels are markers of systemic inflammation. BW-1 = BW from low metallic pads but with some copper, BW-2 = BW from semimetallic pads with no copper, BW-3 = BW from organic brake pads, BW-4 = BW from organic and metallic hybrid pads, KC = keratinocyte-derived chemokine, MIP-2 = macrophage inflammatory protein, PMN = polymorphonuclear neutrophils. Gerlofs-Nijland, B. G. H. Bokkers, H. Sachse, J. J. E. Reijnders, M. Gustafsson, A. J. F. Boere, P. F. H. Fokkens, D. L. A. C. Leseman, K. Augsburg, and F. R. Cassee (2019) Inhalation toxicity profiles of particulate matter: a comparison between brake wear with other sources of emission, *Inhalation Toxicology*, 31:3, 89–98, [10.1080/08958378.2019.1606365](https://doi.org/10.1080/08958378.2019.1606365) by Informa UK Limited, trading as Taylor & Francis.

line (A549).¹⁸⁸ Exposure was carried out at the air–liquid interface; a more physiological model than conventional submerged cell culture. A range of driving and braking behaviors were explored, with the number of repetitions and more abrupt braking modes associating with increased particle mass, number, EC/OC, and metal (Fe, Cu, and Mn) content. While little evidence of overt cytotoxicity was observed,

reduced tightness of the cellular layer, increased oxidative stress (especially with particles produced by full-stop braking) and heightened release of the cytokine IL-8 were observed. The reduced cellular tightness was correlated with particulate Fe, Cu, and Mn content, while inflammation correlated with OC. These associations were, however, based on a limited number of observations and concentrations of Fe, Cu, and Mn

were highly correlated, preventing assessment of their relative contributions to the biological responses.

Focusing on the role of Cu in BWP toxicity, Figliuzzi et al. exposed A549 cells to PM_{2.5} emitted from four different pad/disc combinations.¹⁸⁹ Exposures to 1–500 µg/mL PM (equivalent to 0.0018–70 µg/mL Cu) caused dose-dependent increases in ROS production. Reductions in cell viability were reported at the highest doses (50–500 µg/mL), but not for BWP with mid and low Cu content. Highly Cu-enriched particles disrupted mitochondrial membrane integrity and increased the number of apoptotic cells. Contrastingly, and in agreement with the observations of Gasser et al.,¹⁸⁸ up-regulation of genes relating to oxidative stress and inflammatory responses did not correlate simplistically with particulate Cu content. Barosova et al. used a more sophisticated 3D coculture consisting of A549, monocyte-derived macrophages and dendritic cells challenged with specific size fractions (0.25–1, 1–2, and 2–4 µm) of BWP derived from pad composites of low to moderate metal content.¹⁹⁰ The low metal composite-derived PM elicited no toxicity with any size fraction, whereas the high metallic composite increased IL-8, albeit with no clear dose response, or variation between the collected PM fractions. There were no alterations in intracellular glutathione metabolism, or indications of cell injury. While these findings have been cited as providing evidence of brake dust metal toxicity, it should be noted that the experimental work did not include an analysis of organic chemical species in the PM fractions, an examination of metal handling pathways, or employ selective metal chelators to confirm findings.

Selley et al. compared the toxicity of a composite BWP sample from European buses and trucks with that of a standard reference material DEP.¹⁸¹ Samples varied considerably in their metal/metalloid content, especially for those capable of producing ROS (Fe, Cu, Mn, Mo), indirectly inducing oxidative stress (Zn, Ca) or distinguishing BWP (Ba, Sb, Sn). The DEP contained very little metal, though concentrations of V, As and Ti were broadly equivalent in the two samples. Despite this compositional contrast, both particles induced mitochondrial depolarization, decreased phagocytic capacity, and heightened IL-8 secretion in monocyte-derived macrophages at equivalent doses (4–25 µg/mL). As phagocytosis is a primary mechanism by which macrophages protect the airway from pathogens, the authors hypothesized that both particles could enhance susceptibility to infection. While the similarity in response did not support metals being key drivers of BWP toxicity, responses were inhibited by the metal chelator desferrioxamine for both particles, suggesting that even low concentrations of metals can instigate toxicological effects. This study also highlighted the danger of equating the biological/cellular dose of metals to the measured PM content, as only a subset of the PM-associated metals was shown to dose-dependently accumulate in macrophages.

Several studies have explored BWP toxicity in animal models. We will not, however, cover those that employed asbestos-containing BWP (reviewed by Poland and Duffin¹⁹¹). Gerlofs-Nijland et al. carried out a comprehensive comparative toxicology study in mice, investigating PM from four types of brake pads (low-metal, semimetal, organic, and semimetal–organic hybrid PM) and TRWP from laboratory simulators, as well as DEP, wood-smoke and PM collected in the vicinity of a poultry farm that was enriched with biological components.¹⁹²

Inhalation (9 mg/m³ for up to 6 h) of the high organic and hybrid metal–organic BWP, but not the semimetallic BWP, induced lung inflammation and mild increases in blood leukocytes and fibrinogen. In some cases, the organic-rich BWP had greater inflammatory effect, above TWP, DEP, or wood smoke (Figure 3). While organic-rich brake-wear PM had more marked biological effects than the metallic-BWP, it should be noted that levels of Cu in the organic-PM was actually 4–6 times higher than the metallic brake pads. Furthermore, other pulmonary end points (additional cytotoxicity assays and oxidative stress by glutathione depletion) were unchanged by any of the PM types, and where effects were seen, they were induced by the highest dosing period (6 h exposure). No effects were observed following the shortest dosing period (1.5 h), which the authors suggested was representative of human exposures. This clearly demonstrates that PM with different source profiles elicit markedly different immune responses *in vivo*.

5.3. Toxicity of Tire and Road Wear. Studies of TWP toxicity remain scarce, and due to the methods used to generate the particles, often investigate mixed-source samples from the tire–pavement interface. PM₁₀ generated through wear of studded tires with asphalt or granite mixtures has been shown to induce genotoxicity and mitochondrial depolarization in lung epithelial cells^{193–195} and pro-inflammatory cytokine secretion from macrophages.^{193,196} Compared with BWP, studies of TWP have explored a wider range of toxicological end points. Karlsson et al. performed proteomic profiling of primary human monocyte-derived macrophages exposed to 100 µg/mL PM₁₀ from wear of studded tires against asphalt.¹⁹⁷ The authors did not identify sufficient differentially expressed proteins to perform a statistical pathway analysis but noted that the exposure altered proteins required for inflammation, fibrosis, actin remodelling, and energy metabolism. As well as metals, organic compounds are postulated to mediate TWP toxicity. In a series of experiments with A549 cells, Gualtieri et al. demonstrated that the organic fractions of milled tires (mainly composed of isoprene polymers) induced cytotoxicity, DNA damage and cell cycle arrest following longer exposures (72 h) at doses >60 µg/mL.¹⁹⁸ Subsequent experiments linked toxicity to ROS generation, inhibition of protective Hsp70 activity¹⁹⁹ and redistribution of plasma membrane lipid microdomains.²⁰⁰

In vivo, three studies of responses to TWP exposure have been performed. Gottipolu et al. investigated two types of respirable TWP that leached Fe, Cu, Zn, and Al.²⁰¹ Following pulmonary instillation in rats (~1.5 mg), both PM types increased several markers of pulmonary inflammation, with TWP that contained more Cu and Fe exerting greater effects. Effects were seen at 24 h post exposure and returned to baseline by 1–4 weeks. Neither TWP affected cardiac enzymes. The authors demonstrated that nontire sources of soluble Zn and Cu exerted similar pulmonary responses, albeit the concentration tested was orders of magnitude higher than those leaching from the tires.

A similar approach by Mantecca et al. compared size-fractionated TWP (PM₁₀ and PM_{2.5}) where PAH concentrations were enriched in PM_{2.5}.²⁰² Both PM size fractions induced pulmonary inflammation after instillation into mice (10–200 µg) with greater effects observed with the smaller PM fractions (although responses were variable and often not dose-dependent). Only the high PM_{2.5} doses were associated with overt changes in lung histopathology. The authors suggest

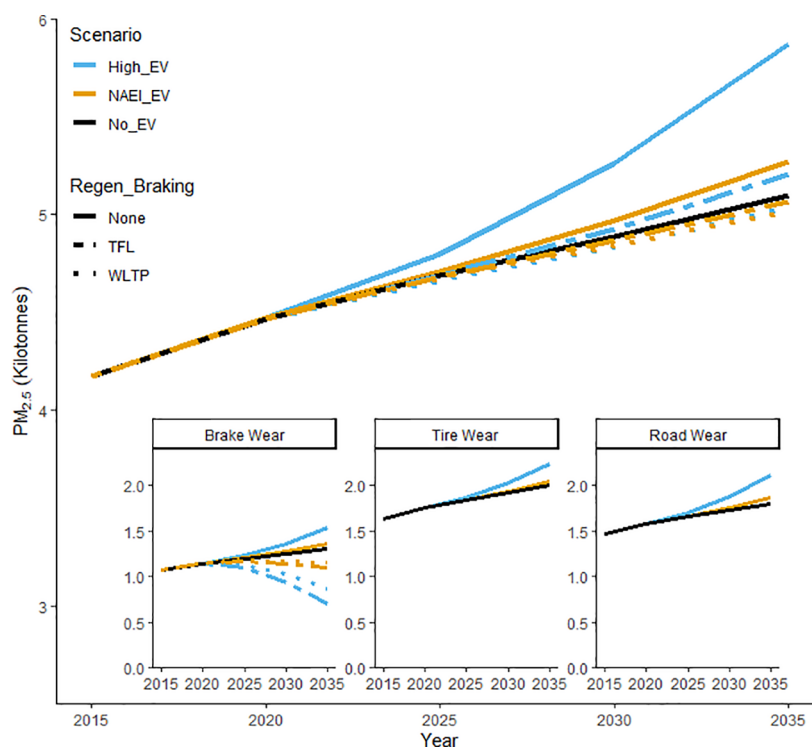


Figure 4. Projected $PM_{2.5}$ brake, tire, and road wear emission estimates for passenger cars in the UK based on changes in DfT (2018) modeled traffic volume and battery electric vehicle uptake (vehicle mass, and regenerative braking). A vehicle mass EF regression approach²⁰⁸ was used to determine the impact of heavier EVs, while reductions from regenerative braking were calculated using brake force simulations for passenger vehicles under the Tfl (dot dashed –65% reduction) and WLTP (small dots –88% reduction) drive cycles. Three scenarios have been considered: “NO_EV”: DfT reference (Scenario 1) traffic projections assuming no electrification of the vehicle fleet; “NAEI_EV”: DfT reference (Scenario 1) traffic projections + UK NAEI EV uptake; “High_EV”: DfT shift to zero emission vehicles (Scenario 7) + high uptake of EVs. DfT: Department for Transport; EF: emission factor; EV: electric vehicles; NAEI National Atmospheric Emissions Inventory; $PM_{2.5}$: particulate matter less than $2.5 \mu m$ in diameter; Tfl: Transport for London; WLTP: Worldwide Harmonized Light Vehicle Test Procedure

a different mechanism of action between the two particle fractions, with macrophages being heavily involved in responses to PM_{10} and cellular cytotoxicity being largely governed by $PM_{2.5}$. The same group compared size fractionated tire particles and similarly sized urban PM collected in Milan, Italy.²⁰³ As before, the $PM_{2.5}$ size fraction provoked an enhanced pulmonary response (inflammation, apoptosis and some signs of adverse histopathology), whereas PM_{10} TWP generally did not. Interestingly, the effects were comparable to, or even greater than urban PM in some assays. The authors calculated that an equivalent dose of $PM_{2.5}$ in humans would be 8 h of $425 \mu g/m^3$; a very high level of exposure, but one that could be reached in an extreme pollution episode in some megacities (although only a small proportion of this would be expected to be derived from TWP). Gerlofs-Nijland et al. performed a comprehensive investigation of the acute toxicity of mixed tire and RWP to that of other sources (see Section 5.2 for details).¹⁹² TWP had a modest capacity to induce pulmonary inflammation, although the relative toxicity varied with end point, with PM from other sources (e.g., BWP, woodstove, particles collected near a poultry farm) often inducing greater responses.

5.4. Overview of Toxicological Studies. Studies show that NEE have the capacity to induce biological actions indicative of potential adverse health effects. In vitro studies have employed several methodological designs, dose ranges, and cell types, although monolayers of lung epithelial cells are the most frequently studied. Both BWP and TWP can exert cytotoxicity at high doses, through different biological

mechanisms with inflammation and oxidative stress being the most studied. Several studies indicate that metals and PAHs are key components driving the toxicological actions observed, consistent with the wider literature on ambient PM. In vivo exposures of NEE in rodents are almost exclusively focused on pulmonary actions, with a handful of studies also examining blood biomarkers. As with in vitro studies, the biological pathways explored/identified are those already established for urban PM and exhaust PM, that is, oxidative stress, coagulation, and inflammation.

In summarizing this literature, we highlight several points. First, numerous studies use PM collected at roadside, and employ measurement of tracers (usually metals) to infer which effects may be linked to nonexhaust PM. However, the proportion of nonexhaust PM within these samples is often not clear and tracers used could originate from other sources. Second, while several studies have considered responses to different particle size fractions, greater attention has focused on coarse and fine PM in NEE, with few studies specifically addressing ultrafine particles. The greater relative surface area of the ultrafine particles, the degree of penetration into the lungs and their ability to translocate into the blood and organs could engender these particles with greater (and different forms of) toxicity. Indeed, there is increasing evidence that ultrafine particles can pass into the circulation and be carried to other organs of the body.²⁰⁴ Maher and co-workers used magnetic quantification and imaging techniques to demonstrate the presence of iron-rich particles (10–150 nm, also containing trace amounts of Pt, Ni, Co, and possibly Cu) in

both the brain²⁰⁵ and the heart²⁰⁶ of cadavers from Mexico and the UK. It has been argued that the rounded morphologies and fused surface textures of these particles may suggest formation at high temperature, possibly from friction on braking. While it is not possible to conclusively prove that these particles have biological actions, cardiac cells containing these particles were found to have histopathological alterations in mitochondrial structure and higher levels of cellular prion protein, consistent with oxidative injury to this organelle. Third, few studies directly compare the biological effects of nonexhaust PM to other PM types. In vitro²⁰⁷ and in vivo¹⁹² studies have shown that some types of nonexhaust PM have greater effects than the same concentration of exhaust PM, but observations were not consistent across different studies or even end points within a given study. In contrast, Kreider et al. referred to unpublished data in rats showing that NEE had no significant effect on lung inflammation, whereas DEP increased the number of inflammatory cells in the lung.¹⁸⁶ Finally, as is conventional in toxicological studies, higher dose ranges are employed to establish effects and thereafter address potential underlying mechanisms. Dose-dependent relationships between particle concentration and effect are found (commonly in in vitro studies, less so in vivo), although significant effects tend to be observed only at higher concentrations, and may not simplistically equate to effects at real world concentrations in humans. Focusing on the two prominent inhalation studies, one study found that short-term exposure to high concentrations (6 h at 9 mg/m³ PM) of brake wear PM induced modest lung inflammation and mild changes in indicators of systemic inflammation.¹⁹² The other study that used prolonged exposures of moderate doses (28 days at 0.1 mg/m³ PM) did not find a mixture of NEE types to have effects on any parameter studied.¹⁸⁶

Overall, based on current research, several forms of NEE have the capacity to induce adverse effects in the lung. These effects are likely linked the ability of NEE to induce inflammation via the induction of oxidative stress, and other forms of biological dysfunction. These studies should stimulate further research, specifically focusing on: (a) impacts other organ systems, (b) toxicity of the ultrafine fraction of NEE, (c) comparative toxicology with other PM sources within the urban environment, and across a range of models, reflective of disease vulnerabilities and (d) at realistic doses. There is also the need to further dissect out the molecular pathways governing adverse outcome pathways linked to components within NEE.

6. FUTURE SCENARIOS AND MITIGATION OF NON-EXHAUST EMISSIONS

6.1. Future Scenarios for Nonexhaust Emissions. The magnitude of future NEE is uncertain due to the unquantified impact of increased vehicle weight, high torque, and regenerative braking associated with electric vehicles (EV),^{208,209} as well as the projected increase in EV mileage due to lower running costs and increased range per battery charge.²¹⁰ This is illustrated graphically in Figure 4, in which brake, tire, and road wear emissions from passenger cars in the UK are forecast to 2035 under different scenarios based on the EMEP/EEA Guidebook EFs¹¹⁴ and vehicle mileage (e.g., activity data).²¹⁰ As shown, the “NAEI EV”²¹¹ scenario (moderate uptake in EVs) and an assumption of no reductions from regenerative braking, would lead to a 3% increase in NEE and the “high EV”²¹⁰ scenario to a 15% increase, compared to

the “No EV” scenario. Regenerative braking would mitigate these effects (~60–90% reduction) by a level dependent on the drive cycle used. These are comparable to an analysis by Hooftman et al., suggesting that regenerative braking reduces brake wear emissions by approximately 66% (despite the heavier mass of EVs),²¹² considering service time of brake linings from internal combustion engines and EVs in urban settings. Beddows and Harrison compared emissions of EV with those of near-equivalent petrol and diesel passenger cars, and assuming 90% regenerative braking on urban and rural road types, there was a small reduction in total nonexhaust particles for the EV despite their greater weight.²⁰⁶ However, regenerative braking does not offset the additional PM₁₀ from heavier EVs on motorways.²⁰⁶

6.2. Mitigation of Nonexhaust Emissions. To reduce NEE, a range of legislative, traffic management, and scientific engineering measures are needed. Reductions in traffic volumes, vehicle speed/velocity, and aggressive driving styles will reduce brake, tire, and resuspension emissions. This could be facilitated by infrastructure design, including road alignment, optimization of road surface texture and traffic signal coordination.²¹³ Trade-offs in the selection of options should, however, be considered. For instance, a low microtexture that decreases tire wear will also reduce friction and hence compromise safety. In contrast macro texture properties might reduce tire wear while also reducing noise and rolling resistance.²¹⁴ Targeting tire/road wear and resuspension emissions is especially important as they increase with vehicle weight and as such, electrification of the fleet. Decreasing the mass of vehicles, especially that of EVs (e.g., by size, design, materials, Li-ion/energy-dense battery technology, smaller EV batteries) would also reduce several nonexhaust sources.^{24,208,209} The use of lighter materials (e.g., carbon composites) may, however, come at the cost of higher environmental impacts during the production phase and lower recyclability of materials.²¹⁵ Introducing taxes on distance traveled and vehicle weight would reduce mileage and discourage heavier vehicles, respectively.²¹⁶ Although these measures may be considered politically complex to implement, they would bring about wider benefits on exhaust/greenhouse gas emissions, road safety, and congestion.

6.2.1. Brake Wear Mitigation. Several US states have set limits on the content of certain heavy metals, (e.g., Cu, Cr, Cd) and asbestos in brake pads has been banned in most regions of the world.⁷⁷ Although these legislations could reduce heavy metals in brake material, they may not successfully reduce overall brake wear emissions if alternative materials have comparable wear properties. It is important to consider specific brake materials (e.g., titanium, aluminum, ceramic, or carbide coatings) with lower wear properties and reduced brake wear emissions. Tungsten carbide and carbon ceramic discs have been shown to reduce PM₁₀ emissions by up to 70%,²¹⁷ while results indicate potential for a European standard car disc brake system to reduce brake wear PM₁₀ by 32–62%.²¹⁸ Future legislation could facilitate the uptake of alternative brake wear materials through financial incentives. Technologies to catch brake wear particles at the source through filtering devices mounted on the brake disc are also under development.^{219,220} It is expected that brake materials will change dramatically in the future and keeping track of composition and tracers is crucial for apportionment studies and identification in environmental samples. Brake wear emissions

and, importantly, the ultrafine fraction can also be mitigated by promoting smoother braking and less aggressive driving.^{45,221}

6.2.2. Tire Wear Mitigation. Tire materials and construction can be optimized toward lower wear but must be balanced against properties related to safety (friction) and environmental (noise, rolling resistance) aspects. Harder rubber might, for instance, reduce wear but decrease friction and increase noise. A tire wear test method to facilitate tire wear marking, enabling customers to choose a lower wearing type, is being investigated within industry²²² and research initiatives (e.g., EU-project LEON-T²²³). While there is already a tread wear rate marking on some tires marketed in the U.S., this does not correlate well to mass loss.⁷⁷ Alternative materials to rubber, a major source of microplastic pollutants, could also contribute to lower emissions. However, the driving forces behind the large effort to find alternatives to synthetic and natural rubber focus on (a) raw material production and transport to reduce greenhouse gas emissions, and (b) agricultural and recycling aspects rather than reducing the wear of the final product. There are also initiatives aimed at collecting TWP while driving.²²⁴ Road surface roughness/condition affect the wear of tires, thus efforts could incorporate adapting surface properties.²²² The driver/car owner can reduce tire wear through driving behavior (e.g., lower acceleration, soft braking/steering maneuvers) and maintaining correct tire inflation pressure and wheel alignment.^{225,226}

6.2.3. Road Wear Mitigation. Road wear is a substantial part of NEE, yet it has not earned much attention outside the studded tire zone. In Sweden the amount of pavement eroded by studded tires was approximated to ~110 000 tons each year, despite the development and use of very wear resistant surfaces since the 1970s.²²⁷ Measures to reduce the use of studded tires through charging schemes (Norway) and prohibition on certain streets (Sweden) have helped to reduce PM concentrations.²²⁸ Other measures focus on designing tires with fewer studs per wheel circumference, which produce less abrasion. Apart from wear resistance, the surface condition affects particle emissions. A study of unstudded tires on pavements in a load simulator concluded that a pavement surface in good condition had very low emissions, while damaged surfaces lead to considerably more.⁹⁵ Compared to smoother pavements with finer aggregates used outside the studded tires zone, wear-resistant pavements are also characterized by higher noise emissions and rolling resistance. The potential impact of the latter on health and fuel consumption means that efforts to reduce them are higher on the mitigation agenda than wear emission reduction. Reducing aggregate size,²²⁹ use of alternative materials in the matrix,²³⁰ and adopting porous pavements²³¹ are some of the measures that have been investigated to reduce noise. Using residual products in the matrix such as scrapped tires and incinerator slag offer an alternative environmental benefit, although current findings are mixed and evaluation of PM emissions is lacking.

6.2.4. Resuspension Mitigation. Resuspension can be mitigated by reducing sources of road dust (Sections 6.2.1–6.2.3) and/or dust suspension. Measures to reduce suspension include reducing surface macro texture (to reduce dust accumulation and facilitate sweeping) and ensuring road surface maintenance to guard against cracks and other dust accumulating damage. In 2015, a European certification test, assessing the PM₁₀/PM_{2.5} efficiency of road sweepers, was established.²³² Road sweeping tests on real roads often show

limited efficiency,^{233,234} explained by the failure of techniques to collect and hold small particles.²³⁵ Techniques can sometimes cause resuspension themselves through brushing without simultaneous dust suppression. Studies combining sweepers and washers²³⁶ or high vacuum sweepers equipped or combined with high-pressure washing (reducing dust load of sub-180 μm particles by up to 92%)^{237,238} have reported more promising results. Another well-tested measure is dust binding (or suppression), normally accomplished through spraying water or hygroscopic solutions (chloride salts, acetates, formates) over the road surface to keep it moist, thereby preventing suspension. Efficiency depends on agent used, dose, concentration, and application criteria, and ranges from a few percent to 70% reduction of resuspended PM₁₀.²³⁹ The effect of dust binding is rather short-lived and dependent on traffic intensity, temperature, and humidity (i.e., parameters affecting how fast the road surface dries).^{112,119} A study in the warm and dry climate of southern Spain showed that a dust binding agent had very little effect, but street washing with high amounts of water was more effective.⁵⁵ Porous pavements have also been shown to conceal dust, thereby contributing to lower resuspension;²⁴⁰ however, the effect diminishes as the pores become clogged, necessitating effective and regular rinsing.²⁴¹ As with all nonexhaust sources, emissions from resuspension are lessened by reduced traffic volumes. Since regulations reducing speed and heavy vehicles in the fleet will reduce resuspension, the current trend toward heavier passenger vehicles is undesirable.

7. DISCUSSION

Nonexhaust particle emissions arising from wear of brakes, tires, and the road surface, and the resuspension of road dust, are unregulated and exceed exhaust emissions in many jurisdictions. Despite this, owing to a lack of epidemiological and toxicological research, we do not have a clear picture of the health risk they pose. This calls for a multidisciplinary research effort to tailor effective and appropriate evidence-based legislation and abatement strategies to protect human health.

Quantitative data on the magnitude of NEE are sparse and highly uncertain owing to different methodological approaches and many complex variables including different vehicle fleets, environmental/meteorological determinants, driving styles (e.g., rural/urban/motorway), operational features (e.g., vehicle speed, acceleration/deceleration), and physical factors (e.g., vehicle mass, brake/tire/road material compositions, and properties). To gain a greater understanding of real-world concentrations to better inform exposure model development and validation for studies on human health, there is a need for a consensus on internationally consistent test setups, to ensure that measurement methods and results are repeatable and reproducible, enabling independent laboratories to assess NEE. To this end, the United Nations Economic Commission for Europe Working Party on Noise and Tyres (Groupe Rapporteur Bruit et Pneumatiques (GRBP)) and industry bodies are developing standardized test procedures to measure brake and tire particle mass and number emissions.^{222,242} At the time of writing, the Particle Measurement Program Working Group has recommended a minimum specification for measurement of brake particle emissions, but not a finalized protocol. Less progress has been made toward standardization of tire particle emissions. When it comes to measuring suspension of or sampling of road dust, methodologies (stationary using a vacuum^{55,243} or wet dust sampler;²⁴⁴ use

of mobile laboratories sampling behind wheels^{245,246}) are at an early stage of development with little or no harmonization.

Another major bottleneck preventing epidemiological studies from fully quantifying the health effects of NEE is the absence of accurate exposure assessment of often highly correlated source components within PM_{2.5} and PM₁₀. Improving exposure estimates will require further investment into intensive sampling campaigns that capture detailed spatial and temporal profiles of NEE by measuring particle mass, size distribution, and chemical composition simultaneously at background locations and near road locations, close to where individuals live and commute.

Current methods to assess short-term exposure can identify nonexhaust sources as *mixtures of pollutants* generated by road dusts, brake wear, and tire wear. Although this indicator is useful for short-term epidemiological studies, precise estimates of *individual nonexhaust sources* are needed. These sources, which display a significant decay in concentrations from the roadside to urban background, may be better apportioned by use of data from personal exposure monitoring, near-roadway monitoring stations, or mobile monitoring that aims to characterize in-vehicle or on-road exposure. The development of effective biomarkers of NEE exposure in blood or urine would also significantly assist, but this is complex given that many of the candidate biomarkers are also associated with other urban stressors, other air pollution sources and poor diet.

For the assessment of long-term exposure, dispersion models have potential to provide PM concentrations attributable to nonexhaust sources. It is noteworthy and informative for future studies, that in most European study areas, elemental PM₁₀ models perform better than PM_{2.5} ones for trace metals.¹⁵⁰ More efforts are however needed to effectively disentangle the correlation with exposure to exhaust PM, as well as other pollutants and constituents thereof, and to generalize the model application more broadly beyond study areas. Future studies utilizing LUR should adopt larger sample sizes and longer monitoring periods to support the development models that can capture temporal variation. Hybrid LUR models that rely on precisely apportioned nonexhaust sources of PM mixtures and can incorporate source-specific predictor variables, such as nonexhaust dispersion model predictions, would be highly valuable for future epidemiological studies. Furthermore, the current changing pattern of vehicle emissions linked to improved abatement of engine exhaust pollutants may provide new opportunities for health effects studies.

To date, human experimental and panel studies have examined acute biochemical, inflammatory, and physiological responses to source specific components of ambient PM through examining responses in highly contrasting micro-environments and the underlying correlations between components and response.^{247,248} Findings from the limited number of epidemiological studies that focused specifically on certain NEE tracer elements or source factors suggest PM Zn and road dust source factors may be associated with acute and chronic cardiovascular outcomes, as well as birth outcomes. These studies have provided some insights, but the results are often and require further validation. There is also an absence of mechanistic work to place the associations observed into causal pathways relevant to disease etiology, progression, or exacerbation.

The toxicological evidence examining the relative hazard of exhaust derived PM versus that from other discrete nonexhaust sources is also limited but evolving. Several forms of NEE have

the capacity to induce adverse effects in the lung.^{192,201} Now attention must turn to other organ systems and establish effects at real-world concentrations. There is a need for defined reference materials, reflecting NEE sources, but a consensus as to what these should be, given the heterogeneity of components within tire and brake wear, and road dust will need to be established first. Such a consensus should consider the need to focus on the toxicity of *all* size fractions of nonexhaust particles. Previous toxicological evidence has examined ambient PM samples collected from roadside, versus urban background locations, and therefore enriched with components of NEE. These studies have often made inferences about NEE components and a range of end points, usually related to the induction of acute inflammatory responses/injury or metabolic response profiles through correlation with PM components associated with NEE.^{184,249} Such approaches have similar limitations to those inherent in epidemiological studies, as many of the chemical components are highly correlated. While this literature, together with the mature toxicological literature on transition metals from the occupational and dietary based research,^{250–253} provides insight into the potential hazard of NEE, difficulties interpreting studies in relation to the relative toxicity of particles from nonexhaust sources should not be underestimated, as there is currently no consensus on the relative toxicity of the major components or sources of PM.^{254,255}

Since many of the metals associated with nonexhaust PM can catalyze oxidation reactions, a consideration of their contribution to the oxidative potential (OP; the capacity of particulate pollution to cause damaging oxidative reactions) of ambient PM is warranted. Indeed, previous studies have indicated that they contribute significantly to this metric and a clear roadside increment in PM_{2.5} and PM₁₀ OP has been demonstrated.²⁵⁶ A greater focus is also required on the nonmetal components of NEE, specifically the organic components derived from brake and tire wear, and trace elements within NEE should also not be excluded for consideration. Furthermore, given the capacity of cells and tissues to regulate metal uptake and store these ions in endogenous chelation proteins, such as ferritin and metallothionein, it is likely that their true hazard will not be reflected in acute exposure models and the accumulative impact of longer-term exposures will need to be addressed.

Currently it is unknown if there are groups of individuals with higher susceptibility to the health effects of NEE and whether this would be similar to that of other air pollutants. Given the high content of Cu, Zn, and Fe in some NEE, individuals with genetic defects in the biological handling of these metals (e.g., individuals with hereditary hemochromatosis: the most common genetic disease in individuals of European ancestry²⁵⁷) would be assumed to be at greater risk. Fe, Zn, and Cu are also essential for microbial growth, and therefore future research should consider changes to the respiratory microbiome in individuals with depressed innate immunity.

Achieving the multidisciplinary elements of a research effort described above is an essential prerequisite to inform policy responses to NEE, such as legislating minimum low emission standard for brakes, tires, and/or road surface materials and/or resuspended road dust. Potential mitigation strategies include development of brake materials with reduced wear properties, identifying alternative materials to rubber for tires, on-vehicle brake/tire wear capture, the use of dust bindings/high vacuum

sweepers combined with high-pressure washing to reduce resuspension, as well as overarching measures of lowering speed, promoting smoother driving behavior, and reducing vehicle mass. Further technical innovation behind such initiatives is encouraged. This must occur in tandem with studies that quantify the efficacy and safety of solutions, ensuring, for example, that new materials to reduce emissions are not associated with equivalent or heightened toxicity that could offset health benefits of this unregulated aspect of vehicular pollution that is forecast to continue to become ever more dominant in future years.

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Notes

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REFERENCES

- (1) Thurston, G. D.; Kipen, H.; Annesi-Maesano, I.; Balmes, J.; Brook, R. D.; Cromar, K.; De Matteis, S.; Forastiere, F.; Forsberg, B.; Frampton, M. W.; Grigg, J.; Heederik, D.; Kelly, F. J.; Kuenzli, N.; Laumbach, R.; Peters, A.; Rajagopalan, S. T.; Rich, D.; Ritz, B.; Samet, J. M.; Sandstrom, T.; Sigsgaard, T.; Sunyer, J.; Brunekreef, B. A joint ERS/ATS policy statement: what constitutes an adverse health effect of air pollution? An analytical framework. *Eur. Respir. J.* **2017**, *49* (1), 1600419.
- (2) W. H. O. WHO global air quality guidelines: Particulate matter (PM_{2.5} and PM₁₀), ozone, nitrogen dioxide, sulfur dioxide and carbon monoxide; World Health Organisation. License: CC BY-NC-SA 3.0 IGO: Geneva, 2021, 2021.
- (3) HEI Special Report 17: Traffic-related air pollution: a critical review of the literature on emissions, exposure and health effects <https://www.healtheffects.org/publication/traffic-related-air-pollution-critical-review-literature-emissions-exposure-and-health> (accessed 2019-3-4).
- (4) Miller, M. R.; Newby, D. E. Air pollution and cardiovascular disease: car sick. *Cardiovasc. Res.* **2019**. DOI: [10.1093/cvr/cvz228](https://doi.org/10.1093/cvr/cvz228)
- (5) Munoz, X.; Barreiro, E.; Bustamante, V.; Lopez-Campos, J. L.; Gonzalez-Barcala, F. J.; Cruz, M. J. Diesel exhausts particles: Their role in increasing the incidence of asthma. Reviewing the evidence of a causal link. *Sci. Total Environ.* **2019**, *652*, 1129–1138.
- (6) Robinson, R. K.; Birrell, M. A.; Adcock, J. J.; Wortley, M. A.; Dubuis, E. D.; Chen, S.; McGilvery, C. M.; Hu, S.; Shaffer, M. S. P.; Bonvini, S. J.; Maher, S. A.; Mudway, I. S.; Porter, A. E.; Carlsten, C.; Tetley, T. D.; Belvisi, M. G. Mechanistic link between diesel exhaust particles and respiratory reflexes. *Journal of Allergy and Clinical Immunology* **2018**, *141* (3), 1074–1084.e9.

- (7) Lund, A. K.; Lucero, J.; Lucas, S.; Madden, M. C.; McDonald, J. D.; Seagrave, J. C.; Knuckles, T. L.; Campen, M. J. Vehicular emissions induce vascular MMP-9 expression and activity associated with endothelin-1-mediated pathways. *Arterioscler Thromb Vasc Biol.* **2009**, *29* (4), 511–7.
- (8) Soares, S. R.; Carvalho-Oliveira, R.; Ramos-Sanchez, E.; Catanozi, S.; da Silva, L. F.; Mauad, T.; Gidlund, M.; Goto, H.; Garcia, M. L. Air pollution and antibodies against modified lipoproteins are associated with atherosclerosis and vascular remodeling in hyperlipemic mice. *Atherosclerosis* **2009**, *207* (2), 368–73.
- (9) Piscitello, A.; Bianco, C.; Casasso, A.; Sethi, R. Non-exhaust traffic emissions: Sources, characterization, and mitigation measures. *Science of The Total Environment* **2021**, *766*, 144440.
- (10) Harrison, R. M.; Allan, J.; Carruthers, D.; Heal, M. R.; Lewis, A. C.; Marner, B.; Murrells, T.; Williams, A. Non-exhaust vehicle emissions of particulate matter and VOC from road traffic: A review. *Atmos. Environ.* **2021**, *262*, 118592.
- (11) Jeong, C.-H.; Wang, J. M.; Hilker, N.; Debosz, J.; Sofowote, U.; Su, Y.; Noble, M.; Healy, R. M.; Munoz, T.; Dabek-Zlotorzynska, E.; Celo, V.; White, L.; Audette, C.; Herod, D.; Evans, G. J. Temporal and spatial variability of traffic-related PM_{2.5} sources: Comparison of exhaust and non-exhaust emissions. *Atmos. Environ.* **2019**, *198*, 55–69.
- (12) Air Quality Expert Group (AQEG) *Non-Exhaust Emissions from Road Traffic*, **2019**.
- (13) Kelly, F. J. Oxidative stress: its role in air pollution and adverse health effects. *Occupational and Environmental Medicine* **2003**, *60* (8), 612–616.
- (14) Garg, B. D.; Cadle, S. H.; Mulawa, P. A.; Groblicki, P. J.; Laroo, C.; Parr, G. A. Brake Wear Particulate Matter Emissions. *Environ. Sci. Technol.* **43**, (21), 4463–4469.
- (15) Mathissen, M.; Scheer, V.; Vogt, R.; Benter, T. Investigation on the potential generation of ultrafine particles from the tire–road interface. *Atmos. Environ.* **2011**, *45* (34), 6172–6179.
- (16) Amato, F.; Cassee, F. R.; Denier van der Gon, H. A.; Gehrig, R.; Gustafsson, M.; Hafner, W.; Harrison, R. M.; Jozwicka, M.; Kelly, F. J.; Moreno, T.; Prevot, A. S.; Schaap, M.; Sunyer, J.; Querol, X. Urban air quality: the challenge of traffic non-exhaust emissions. *J. Hazard Mater.* **2014**, *275*, 31–6.
- (17) Gulia, S.; Goyal, P.; Goyal, S. K.; Kumar, R. Re-suspension of road dust: contribution, assessment and control through dust suppressants—a review. *International Journal of Environmental Science and Technology* **2019**, *16* (3), 1717–1728.
- (18) Hagino, H.; Oyama, M.; Sasaki, S. Laboratory Testing of Airborne Brake Wear Particle Emissions using a Dynamometer System under Urban City Driving Cycles. *Atmospheric environment* **2016**, *131*, 269–278.
- (19) Sanders, P. G.; Xu, N.; Dalka, T. M.; Maricq, M. M. Airborne brake wear debris: size distributions, composition, and a comparison of dynamometer and vehicle tests. *Environ. Sci. Technol.* **2003**, *37* (18), 4060–4069.
- (20) Gustafsson, M.; Blomqvist, G.; Gudmundsson, A.; Dahl, A.; Jonsson, P.; Swietlicki, E. Factors influencing PM₁₀ emissions from road pavement wear. *Atmos. Environ.* **2009**, *43* (31), 4699–4702.
- (21) Alves, C.; Evtugina, M.; Vicente, A.; Conca, E.; Amato, F. Organic profiles of brake wear particles. *Atmospheric research.* **2021**, *255*, 105557.
- (22) Mathissen, M.; Grigoratos, T.; Lähde, T.; Vogt, R. Brake Wear Particle Emissions of a Passenger Car Measured on a Chassis Dynamometer. *Atmosphere* **2019**, *10*, 556.
- (23) Farwick zum Hagen, F. H.; Mathissen, M.; Grabiec, T.; Hennicke, T.; Rettig, M.; Grochowicz, J.; Vogt, R.; Benter, T. Study of Brake Wear Particle Emissions: Impact of Braking and Cruising Conditions. *Environ. Sci. Technol.* **2019**, *53* (9), 5143–5150.
- (24) Oroumiyeh, F.; Zhu, Y. Brake and tire particles measured from on-road vehicles: Effects of vehicle mass and braking intensity. *Atmospheric Environment: X* **2021**, *12*, 100121.
- (25) Beji, A.; Deboudt, K.; Khardi, S.; Muresan, B.; Lumière, L. Determinants of rear-of-wheel and tire-road wear particle emissions by light-duty vehicles using on-road and test track experiments. *Atmospheric Pollution Research* **2021**, *12* (3), 278–291.
- (26) Hicks, W.; Beevers, S.; Tremper, A. H.; Stewart, G.; Priestman, M.; Kelly, F. J.; Lanoisellé, M.; Lowry, D.; Green, D. C. Quantification of Non-Exhaust Particulate Matter Traffic Emissions and the Impact of COVID-19 Lockdown at London Marylebone Road. *Atmosphere* **2021**, *12* (2), 190.
- (27) Harrison, R. M.; Jones, A. M.; Gietl, J.; Yin, J.; Green, D. C. Estimation of the contributions of brake dust, tire wear, and resuspension to nonexhaust traffic particles derived from atmospheric measurements. *Environ. Sci. Technol.* **2012**, *46* (12), 6523–9.
- (28) Charron, A.; Polo-Rehn, L.; Besombes, J.-L.; Golly, B.; Buisson, C.; Chanut, H.; Marchand, N.; Guillaud, G.; Jaffredo, J.-L. Identification and quantification of particulate tracers of exhaust and non-exhaust vehicle emissions. *Atmospheric Chemistry and Physics* **2019**, *19* (7), 5187–5207.
- (29) Harrison, R. M.; Beddows, D. C. S.; Dall'Osto, M. PMF analysis of wide-range particle size spectra collected on a major highway. *Environ. Sci. Technol.* **2011**, *45* (13), 5522–5528.
- (30) Crilley, L. R.; Lucarelli, F.; Bloss, W. J.; Harrison, R. M.; Beddows, D. C.; Calzolari, G.; Nava, S.; Valli, G.; Bernardoni, V.; Vecchi, R. Source apportionment of fine and coarse particles at a roadside and urban background site in London during the 2012 summer ClearfLo campaign. *Environ. Pollut.* **2017**, *220*, 766–778.
- (31) 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. Advanced source apportionment of size-resolved trace elements at multiple sites in London during winter. *Atmospheric Chemistry and Physics* **2015**, *15* (19), 11291–11309.
- (32) Wahlström, J.; Söderberg, A.; Olofsson, U. Simulation of airborne wear particles from disc brakes. In *SAE Technical Papers*, **2009**.
- (33) Nosko, O.; Olofsson, U. Effective density of airborne wear particles from car brake materials. In *J. Aerosol Sci.*; Elsevier Ltd, **2017**; Vol. *107*, pp 94–106.
- (34) Hagino, H.; Oyama, M.; Sasaki, S. Laboratory testing of airborne brake wear particle emissions using a dynamometer system under urban city driving cycles. In *Atmos. Environ.*; Elsevier Ltd, **2016**; Vol. *131*, pp 269–278.
- (35) Iijima, A.; Sato, K.; Yano, K.; Tago, H.; Kato, M.; Kimura, H.; Furuta, N. Particle size and composition distribution analysis of automotive brake abrasion dusts for the evaluation of antimony sources of airborne particulate matter. *Atmos. Environ.* **2007**, *41*, 4908–4919.
- (36) Iijima, A.; Sato, K.; Yano, K.; Kato, M.; Kozawa, K.; Furuta, N. Emission factor for antimony in brake abrasion dusts as one of the major atmospheric antimony sources. *Environ. Sci. Technol.* **2008**, *42*, 2937–2942.
- (37) Kukutschová, J.; Moravec, P.; Tomášek, V.; Matějka, V.; Smolík, J.; Schwarz, J.; Seidlerová, J.; Šafářová, K.; Filip, P. On airborne nano/micro-sized wear particles released from low-metallic automotive brakes. In *Environ. Pollut.*; Elsevier: **2011**; Vol. *159*, pp 998–1006.
- (38) Kwak, J. H.; Kim, H.; Lee, J.; Lee, S. Characterization of non-exhaust coarse and fine particles from on-road driving and laboratory measurements. In *Sci. Total Environ.*; Elsevier, **2013**; Vol. *458–460*, pp 273–282.
- (39) Park, J.; Joo, B.; Seo, H.; Song, W.; Lee, J. J.; Lee, W. K.; Jang, H. Analysis of wear induced particle emissions from brake pads during the worldwide harmonized light vehicles test procedure (WLTP). In *Wear*; Elsevier B.V., **2021**; Vol. *466–467*, p 203539.
- (40) Alemani, M.; Wahlström, J.; Olofsson, U. On the influence of car brake system parameters on particulate matter emissions. In *Wear*; Elsevier B.V., **2018**; Vol. *396–397*, pp 67–74.

- (41) Alemani, M.; Nosko, O.; Metinoz, I.; Olofsson, U. A study on emission of airborne wear particles from car brake friction pairs. *SAE International Journal of Materials and Manufacturing* **2016**, *9*, 147–157.
- (42) Farwick zum Hagen, F. H.; Mathissen, M.; Grabiec, T.; Hennicke, T.; Rettig, M.; Grochowicz, J.; Vogt, R.; Benter, T. On-road vehicle measurements of brake wear particle emissions. In *Atmos. Environ.*; Elsevier, 2019; Vol. 217, p 116943.
- (43) Garg, B. D.; Cadle, S. H.; Mulawa, P. A.; Groblicki, P. J.; Laroo, C.; Parr, G. A. Brake Wear Particulate Matter Emissions. *Environ. Sci. Technol.* **2000**, *34* (21), 4463–4469.
- (44) Niemann, H.; Winner, H.; Asbach, C.; Kaminski, H.; Frentz, G.; Milczarek, R. Influence of disc temperature on ultrafine, fine, and coarse particle emissions of passenger car disc brakes with organic and inorganic pad binder materials. In *Atmosphere*, **2020**; Vol. 11.1060
- (45) Nosko, O.; Olofsson, U. Quantification of ultrafine airborne particulate matter generated by the wear of car brake materials. In *Wear*; Elsevier B.V.: 2017; Vol. 374–375, pp 92–96.
- (46) Chandra Verma, P.; Menapace, L.; Bonfanti, A.; Ciudin, R.; Gialanella, S.; Straffellini, G. Braking pad-disc system: Wear mechanisms and formation of wear fragments. In *Wear*; Elsevier, 2015; Vol. 322–323, pp 251–258.
- (47) Liati, A.; Schreiber, D.; Lugovyy, D.; Gramstat, S.; Dimopoulos Eggenschwiler, P. Airborne particulate matter emissions from vehicle brakes in micro- and nano-scales: Morphology and chemistry by electron microscopy. In *Atmos. Environ.*; Elsevier, 2019; Vol. 212, pp 281–289.
- (48) Verma, P. C.; Alemani, M.; Gialanella, S.; Lutterotti, L.; Olofsson, U.; Straffellini, G. Wear debris from brake system materials: A multi-analytical characterization approach. In *Tribology International*; Elsevier, 2016; Vol. 94, pp 249–259.
- (49) Wahlström, J.; Olander, L.; Olofsson, U. Size, shape, and elemental composition of airborne wear particles from disc brake materials. *Tribol. Lett.* **2010**, *38*, 15–24.
- (50) Hulskotte, J. H. J.; Roskam, G. D.; Denier van der Gon, H. A. C. Elemental composition of current automotive braking materials and derived air emission factors. In *Atmos. Environ.*; Elsevier Ltd, 2014; Vol. 99, pp 436–445.
- (51) Figi, R.; Nagel, O.; Tuchschnid, M.; Lienemann, P.; Gfeller, U.; Bukowiecki, N. Quantitative analysis of heavy metals in automotive brake linings: A comparison between wet-chemistry based analysis and in-situ screening with a handheld X-ray fluorescence spectrometer. In *Anal. Chim. Acta*; Elsevier, 2010; Vol. 676, pp 46–52.
- (52) Hjortenkrans, D. *Road Traffic Metals—Sources and Emissions*; University of Kalmar, 2008.
- (53) Sethupathi, P. B.; Chandradass, J.; Saibalaji, M. A. Comparative study of disc brake pads sold in Indian market — Impact on safety and environmental aspects. In *Environmental Technology and Innovation*; Elsevier B.V., 2021; Vol. 21, p 101245.
- (54) Vontorová, J.; Dobiáš, V.; Mohyla, P. Utilization of GDOES for the study of friction layers formed on the surface of brake discs during the friction process. *Chemical Papers* **2017**, *71*, 1507–1514.
- (55) Amato, F.; Favez, O.; Pandolfi, M.; Alastuey, A.; Querol, X.; Moukhtar, S.; Bruge, B.; Verlhac, S.; Orza, J. A. G.; Bonnaire, N.; Le Priol, T.; Petit, J. F.; Sciare, J. Traffic induced particle resuspension in Paris: Emission factors and source contributions. *Atmos. Environ.* **2016**, *129*, 114–124.
- (56) Pakbin, P.; Ning, Z.; Shafer, M. M.; Schauer, J. J.; Sioutas, C. Seasonal and Spatial Coarse Particle Elemental Concentrations in the Los Angeles Area. In *Aerosol Sci. Technol.*; Taylor & Francis Group, 2011; Vol. 45, pp 949–963.
- (57) Młeczek, P.; Borowiak, K.; Budka, A.; Szostek, M.; Niedzielski, P. Possible sources of rare earth elements near different classes of road in Poland and their phytoextraction to herbaceous plant species. *Environ. Res.*, **2021**; Vol. 193.110580
- (58) Hussain, S.; Abdul Hamid, M. K.; Mat Lazim, A. R.; Abu Bakar, A. R. Brake wear particle size and shape analysis of non-asbestos organic (NAO) and semi metallic brake pad. *Jurnal Teknologi* **2014**, *71*, 129–134.
- (59) Malachova, K.; Kukutschova, J.; Rybkova, Z.; Sezimova, H.; Placha, D.; Cabanova, K.; Filip, P. Toxicity and mutagenicity of low-metallic automotive brake pad materials. In *Ecotoxicology and Environmental Safety*; Elsevier, 2016; Vol. 131, pp 37–44.
- (60) Menapace, C.; Leonardi, M.; Secchi, M.; Bonfanti, A.; Gialanella, S.; Straffellini, G. Thermal behavior of a phenolic resin for brake pad manufacturing. In *J. Therm. Anal. Calorim.*; Springer International Publishing, 2019; Vol. 137, pp 759–766.
- (61) Almeida, S. M.; Manousakas, M.; Diapouli, E.; Kertesz, Z.; Samek, L.; Hristova, E.; Šega, K.; Alvarez, R. P.; Belis, C. A.; Eleftheriadis, K.; Albania; Civici, N.; Radic, R.; Vukic, L.; Hristova, E.; Veleva, B.; Šega, K.; Bešlić, I.; Davila, S.; Godec, R.; Manousakas, M.; Diapouli, E.; Vratolis, S.; Eleftheriadis, K.; Kertesz, Z.; Belis, C. A.; Bernatonis, M.; Djukanovic, G.; Jancic, D.; Samek, L.; Furman, L.; Stegowski, Z.; Almeida, S. M.; Galinha, C.; Balan, V.; Nikolovska, L.; Stefanovska, A.; Radenkovic, M.; Knežević, J.; Banu Oztas, N.; Cantay, E.; Turchenko, D. V.; Abdullaev, S.; Padilla Alvarez, R.; Karydas, A. G. Ambient particulate matter source apportionment using receptor modelling in European and Central Asia urban areas. In *Environ. Pollut.* **2020**; Vol. 266.115199
- (62) Jeong, C. H.; Traub, A.; Huang, A.; Hilker, N.; Wang, J. M.; Herod, D.; Dabek-Zlotorzynska, E.; Celó, V.; Evans, G. J. Long-term analysis of PM_{2.5} from 2004 to 2017 in Toronto: Composition, sources, and oxidative potential. *Environ. Pollut.* **2020**, *263*, 114652.
- (63) Schauer, J. J.; Lough, G. C.; Shafer, M. M.; Christensen, W. F.; Arndt, M. F.; DeMinter, J. T.; Park, J.-S. Characterization of metals emitted from motor vehicles. *Research Report (Health Effects Institute)* **2006**, *133*, 1–88.
- (64) Oroumiyeh, F.; Jerrett, M.; Del Rosario, I.; Lipsitt, J.; Liu, J.; Paulson, S. E.; Ritz, B.; Schauer, J. J.; Shafer, M. M.; Shen, J.; Weichenthal, S.; Banerjee, S.; Zhu, Y. Elemental composition of fine and coarse particles across the greater Los Angeles area: Spatial variation and contributing sources. *Environ. Pollut.* **2021**, 118356.
- (65) Bukowiecki, N.; Lienemann, P.; Hill, M.; Figi, R.; Richard, A.; Furger, M.; Rickers, K.; Falkenberg, G.; Zhao, Y.; Cliff, S. S.; Prevot, A. S. H.; Baltensperger, U.; Buchmann, B.; Gehrig, R. Real-World Emission Factors for Antimony and Other Brake Wear Related Trace Elements: Size-Segregated Values for Light and Heavy Duty Vehicles. In *Environ. Sci. Technol.*; American Chemical Society, 2009; Vol. 43, pp 8072–8078.
- (66) Duong, T. T. T.; Lee, B. K. Determining contamination level of heavy metals in road dust from busy traffic areas with different characteristics. In *Journal of Environmental Management*; Elsevier Ltd, 2011; Vol. 92, pp 554–562.
- (67) Wang, J. M.; Jeong, C. H.; Hilker, N.; Healy, R. M.; Sofowote, U.; Debosz, J.; Su, Y.; Munoz, A.; Evans, G. J. Quantifying metal emissions from vehicular traffic using real world emission factors. In *Environ. Pollut.*; Elsevier Ltd, 2021; Vol. 268, p 115805.
- (68) Cesari, D.; Amato, F.; Pandolfi, M.; Alastuey, A.; Querol, X.; Contini, D. An inter-comparison of PM₁₀ source apportionment using PCA and PMF receptor models in three European sites. In *Environ. Sci. Pollut. Res.*; Environmental Science and Pollution Research, 2016; Vol. 23, pp 15133–15148.
- (69) Querol, X.; Viana, M.; Alastuey, A.; Amato, F.; Moreno, T.; Castillo, S.; Pey, J.; de la Rosa, J.; Sánchez de la Campa, A.; Artiñano, B.; Salvador, P.; García Dos Santos, S.; Fernández-Patier, R.; Moreno-Grau, S.; Negral, L.; Minguillón, M. C.; Monfort, E.; Gil, J. I.; Inza, A.; Ortega, L. A.; Santamaría, J. M.; Zabalza, J. Source origin of trace elements in PM from regional background, urban and industrial sites of Spain. *Atmos. Environ.* **2007**, *41*, 7219–7231.
- (70) Dong, S.; Ochoa Gonzalez, R.; Harrison, R. M.; Green, D.; North, R.; Fowler, G.; Weiss, D. Isotopic signatures suggest important contributions from recycled gasoline, road dust and non-exhaust traffic sources for copper, zinc and lead in PM₁₀ in London, United Kingdom. *Atmos. Environ.* **2017**, *165*, 88–98.

- (71) Amato, F. *Non-Exhaust Emissions: An Urban Air Quality Problem for Public Health*; Elsevier Science & Technology: Saint Louis, MO, 2018.
- (72) Adachi, K.; Tainosho, Y. Characterization of heavy metal particles embedded in tire dust. *Environ. Int.* **2004**, *30* (8), 1009–1017.
- (73) Kreider, M. L.; Panko, J. M.; McAtee, B. L.; Sweet, L. I.; Finley, B. L. Physical and chemical characterization of tire-related particles: Comparison of particles generated using different methodologies. *Sci. Total Environ.* **2010**, *408* (3), 652–659.
- (74) Panko, J.; Hitchcock, K.; Fuller, G.; Green, D. Evaluation of Tire Wear Contribution to PM_{2.5} in Urban Environments. *Atmosphere* **2019**, *10*, 99.
- (75) Panko, J. M.; Chu, J.; Kreider, M. L.; Unice, K. M. Measurement of airborne concentrations of tire and road wear particles in urban and rural areas of France, Japan, and the United States. *Atmos. Environ.* **2013**, *72*, 192–199.
- (76) Kim, G.; Lee, S. Characteristics of Tire Wear Particles Generated by a Tire Simulator under Various Driving Conditions. *Environ. Sci. Technol.* **2018**, *52* (21), 12153–12161.
- (77) Grigoratos, T.; Gustafsson, M.; Eriksson, O.; Martini, G. Experimental investigation of tread wear and particle emission from tyres with different treadwear marking. *Atmos. Environ.* **2018**, *182*, 200–212.
- (78) Alves, C. A.; Vicente, A. M. P.; Calvo, A. I.; Baumgardner, D.; Amato, F.; Querol, X.; Pio, C.; Gustafsson, M. Physical and chemical properties of non-exhaust particles generated from wear between pavements and tyres. In *Atmos. Environ.*; Elsevier Ltd, 2020; Vol. 224, p 117252.
- (79) Foitzik, M. J.; Unrau, H. J.; Gauterin, F.; Dörnhöfer, J.; Koch, T. Investigation of ultra fine particulate matter emission of rubber tires. In *Wear*; Elsevier B.V., 2018; Vol. 394–395, pp 87–95.
- (80) Dall'Osto, M.; Beddows, D. C. S.; Gietl, J. K.; Olatunbosun, O. A.; Yang, X.; Harrison, R. M. Characteristics of tyre dust in polluted air: Studies by single particle mass spectrometry (ATOFMS). In *Atmos. Environ.*; Elsevier Ltd, 2014; Vol. 94, pp 224–230.
- (81) Sjödin, Å.; Ferm, M.; Björk, A.; Rahmberg, M.; Gudmundsson, E.; Swietlicki, E.; Johansson, C.; Gustafsson, M.; Blomqvist, G. In *Wear particles from road traffic—A field, laboratory and modelling study*, Gotenberg, 2010; p 101.
- (82) Beji, A.; Deboudt, K.; Khardi, S.; Muresan, B.; Flament, P.; Fourmentin, M.; Lumière, L. Non-exhaust particle emissions under various driving conditions: Implications for sustainable mobility. In *Transportation Research Part D: Transport and Environment*; Elsevier, 2020; Vol. 81, p 102290.
- (83) Han, S.; Jung, Y. W. A study on the characteristics of silt loading on paved roads in the Seoul metropolitan area using a mobile monitoring system. *J. Air Waste Manag Assoc.* **2012**, *62* (7), 846–862.
- (84) Etyemezian, V. Vehicle-based road dust emission measurement: I—methods and calibration. *Atmos. Environ.* **2003**, *37* (32), 4559–4571.
- (85) Pirjola, L.; Kupiainen, K. J.; Perhoniemi, P.; Tervahattu, H.; Vesala, H. Non-exhaust emission measurement system of the mobile laboratory SNIFFER. *Atmos. Environ.* **2009**, *43* (31), 4703–4713.
- (86) Lee, S.; Kwak, J.; Kim, H.; Lee, J. Properties of roadway particles from interaction between the tire and road pavement. *International Journal of Automotive Technology* **2013**, *14* (1), 163–173.
- (87) Dahl, A.; Gharibi, A.; Swietlicki, E.; Gudmundsson, A.; Bohgard, M.; Ljungman, A.; Blomqvist, G.; Gustafsson, M. Traffic-generated emissions of ultrafine particles from pavement–tire interface. In *Atmos. Environ.*; Pergamon, 2006; Vol. 40, pp 1314–1323.
- (88) Gehrig, R. C. H.; Hofer, P. *Contributions of Road Traffic to Ambient PM₁₀ and PM_{2.5} Concentrations*; Swiss Federal Laboratories for Materials Testing and Research: Dübendorf, 2001.
- (89) Wagner, S.; Hüffer, T.; Klöckner, P.; Wehrhahn, M.; Hofmann, T.; Reemtsma, T. Tire wear particles in the aquatic environment - A review on generation, analysis, occurrence, fate and effects. *Water Res.* **2018**, *139*, 83–100.
- (90) Camatini, M.; Crosta, G.; Dolukhanyan, T.; Sung, C.; Giuliani, G.; Corbetta, G.; Cencetti, S.; Regazzoni, C. Microcharacterization and identification of tire debris in heterogeneous laboratory and environmental specimens. *Mater. Charact.* **2001**, *46*, 271–283.
- (91) Kumata, H.; Sanada, Y.; Takada, H.; Ueno, T. Historical Trends of N-Cyclohexyl-2-benzothiazolamine, 2-(4-Morpholinyl)-benzothiazole, and Other Anthropogenic Contaminants in the Urban Reservoir Sediment Core. *Environ. Sci. Technol.* **2000**, *34* (2), 246–253.
- (92) Harrison, R. M.; Jones, A. M.; Gietl, J.; Yin, J.; Green, D. C. Estimation of the contributions of brake dust, tire wear, and resuspension to nonexhaust traffic particles derived from atmospheric measurements. *Environ. Sci. Technol.* **2012**, *46* (12), 6523–6529.
- (93) Unice, K. M.; Kreider, M. L.; Panko, J. M. Use of a Deuterated Internal Standard with Pyrolysis-GC/MS Dimeric Marker Analysis to Quantify Tire Tread Particles in the Environment. *Int. J. Environ. Res. Public Health* **2012**, *9* (11), 4033.
- (94) ISO, ISO/TS 21396:2017 Rubber — Determination of mass concentration of tire and road wear particles (TRWP) in soil and sediments—Pyrolysis-GC/MS method, 2017.
- (95) Gehrig, R.; Zeyer, K.; Bukowiecki, N.; Lienemann, P.; Poulidakos, L. D.; Furger, M.; Buchmann, B. Mobile load simulators - A tool to distinguish between the emissions due to abrasion and resuspension of PM₁₀ from road surfaces. *Atmos. Environ.* **2010**, *44* (38), 4937–4943.
- (96) Kupiainen, K. J.; Tervahattu, H.; Räisänen, M.; Mäkelä, T.; Aurela, M.; Hillamo, R. Size and composition of airborne particles from pavement wear, tires, and traction sanding. *Environ. Sci. Technol.* **2005**, *39* (3), 699–706.
- (97) Gustafsson, M.; Blomqvist, G.; Gudmundsson, A.; Dahl, A.; Swietlicki, E.; Bohgard, M.; Lindbom, J.; Ljungman, A. Properties and toxicological effects of particles from the interaction between tyres, road pavement and winter traction material. *Science of The Total Environment* **2008**, *393* (2), 226–240.
- (98) Kupiainen, K.; Tervahattu, H.; Räisänen, M. Experimental studies about the impact of traction sand on urban road dust composition. *Science of The Total Environment* **2003**, *308* (1–3), 175–184.
- (99) Gustafsson, M.; Johansson, C. Road pavements and PM₁₀. Summary of the results of research funded by the Swedish Transport Administration on how the properties of road pavements influence emissions and the properties of wear particles. In *Trafikverket, Report 2012:241*, December, 2012; ISBN: 978–91–7467–432–3.
- (100) Fauser, P.; Tjell, J. C.; Mosbaek, H.; Pilegaard, K. TIRE-TREAD AND BITUMEN PARTICLE CONCENTRATIONS IN AEROSOL AND SOIL SAMPLES. *Petroleum Science and Technology* **2002**, *20* (1–2), 127–141.
- (101) Thorpe, A.; Harrison, R. M. Sources and properties of non-exhaust particulate matter from road traffic: A review. *Sci. Total Environ.* **2008**, *400* (1–3), 270–282.
- (102) Casotti Rienda, I.; Alves, C. A. Road dust resuspension: A review. *Atmos. Res.* **2021**, *261*, 105740.
- (103) Alves, C. A.; Evtuygina, M.; Vicente, A. M. P.; Vicente, E. D.; Nunes, T. V.; Silva, P. M. A.; Duarte, M. A. C.; Pio, C. A.; Amato, F.; Querol, X. Chemical profiling of PM₁₀ from urban road dust. *Science of The Total Environment* **2018**, *634*, 41–51.
- (104) Candéias, C.; Vicente, E.; Tomé, M.; Rocha, F.; Ávila, P.; Alves, C. Geochemical, mineralogical and morphological characterisation of road dust and associated health risks. *International Journal of Environmental Research and Public Health* **2020**, *17* (5), 1563.
- (105) Gustafsson, M.; Blomqvist, G.; Järskog, I.; Lundberg, J.; Janhäll, S.; Elmgren, M.; Johansson, C.; Norman, M.; Silvergren, S. Road dust load dynamics and influencing factors for six winter seasons in Stockholm, Sweden. *Atmospheric Environment: X* **2019**, *2*.
- (106) Blomqvist, G.; Gustafsson, M.; Lundberg, T. Road surface dust load is dependent on road surface macro texture. In *European Aerosol Conference*, Prague, Czech Republic, 2013.
- (107) Amato, F.; Schaap, M.; Denier van der Gon, H. A. C.; Pandolfi, M.; Alastuey, A.; Keuken, M.; Querol, X. Effect of rain

events on the mobility of road dust load in two Dutch and Spanish roads. *Atmos. Environ.* **2012**, *46*, 352–358.

(108) China, S.; James, D. E. Influence of pavement macrotexture on PM₁₀ emissions from paved roads: A controlled study. *Atmos. Environ.* **2012**, *46*, 313–326.

(109) Amato, F.; Karanasiou, A.; Cordoba, P.; Alastuey, A.; Moreno, T.; Lucarelli, F.; Nava, S.; Calzolari, G.; Querol, X. Effects of road dust suppressants on PM levels in a mediterranean urban area. *Environ. Sci. Technol.* **2014**, *48* (14), 8069–8077.

(110) Singh, V.; Biswal, A.; Kesarkar, A. P.; Mor, S.; Ravindra, K. High resolution vehicular PM₁₀ emissions over megacity Delhi: Relative contributions of exhaust and non-exhaust sources. *Sci. Total Environ.* **2020**, 699.134273

(111) Kupiainen, K.; Tervahattu, H. The effect of traction sanding on urban suspended particles in Finland. *Environmental Monitoring and Assessment* **2004**, *93* (1–3), 287–300.

(112) Denby, B. R.; Sundvor, I.; Johansson, C.; Pirjola, L.; Ketzler, M.; Norman, M.; Kupiainen, K.; Gustafsson, M.; Blomqvist, G.; Kauhaniemi, M.; Omstedt, G. A coupled road dust and surface moisture model to predict non-exhaust road traffic induced particle emissions (NORTRIP). Part 2: Surface moisture and salt impact modelling. *Atmos. Environ.* **2013**, *47*, 485–503.

(113) USEPA. *Brake and Tire Wear Emissions from Onroad Vehicles in MOVES3 Brake and Tire Wear Emissions*. 2020; Vol. 2020, p 48

(114) Ntziachristos, L.; Boulter, P. *EMEP/EEA air pollutant emission inventory guidebook 2019*, 2019.

(115) Amato, F. *Non-Exhaust Emissions. An Urban Air Quality Problem for Public Health. Impact and Mitigation Measures*; Academic Press, 2018.

(116) Denby, B.; Kupiainen, K.; Gustafsson, M. Review of Road Dust Emissions. In *Non-Exhaust Emissions - An Urban Air Quality Problem for Public Health*; Amato, F., Ed.; Academic Press, 2018.

(117) Amato, F.; Bedogni, M.; Padoan, E.; Querol, X.; Ealo, M.; Rivas, I. Characterization of road dust emissions in milan: Impact of vehicle fleet speed. *Aerosol Air Qual. Res.* **2017**, *17* (10), 2438–2449.

(118) USEPA. AP42 Emission factor Documentation for AP-42, Section 13.2.1 Paved Roads - January 2011, U.S. Environmental Protection Agency: Washington, DC, 2011.

(119) Denby, B. R.; Sundvor, I.; Johansson, C.; Pirjola, L.; Ketzler, M.; Norman, M.; Kupiainen, K.; Gustafsson, M.; Blomqvist, G.; Omstedt, G. A coupled road dust and surface moisture model to predict non-exhaust road traffic induced particle emissions (NORTRIP). Part 1: Road dust loading and suspension modelling. *Atmos. Environ.* **2013**, *47*, 283–300.

(120) Samoli, E.; Atkinson, R. W.; Analitis, A.; Fuller, G. W.; Green, D. C.; Mudway, I.; Anderson, H. R.; Kelly, F. J. Associations of short-term exposure to traffic-related air pollution with cardiovascular and respiratory hospital admissions in London, UK. *Occup Environ. Med.* **2016**, *73* (5), 300–7.

(121) Zhang, J.; Zhang, X.; Wu, L.; Wang, T.; Zhao, J.; Zhang, Y.; Men, Z.; Mao, H. Occurrence of benzothiazole and its derivatives in tire wear, road dust, and roadside soil. *Chemosphere* **2018**, *201*, 310–317.

(122) Unice, K. M.; Kreider, M. L.; Panko, J. M. Use of a Deuterated Internal Standard with Pyrolysis-GC/MS Dimeric Marker Analysis to Quantify Tire Tread Particles in the Environment. *International Journal of Environmental Research and Public Health* **2012**, *9* (11), 4033–4055.

(123) Krall, J. R.; Ladva, C. N.; Russell, A. G.; Golan, R.; Peng, X.; Shi, G.; Greenwald, R.; Raysoni, A. U.; Waller, L. A.; Sarnat, J. A. Source-specific pollution exposure and associations with pulmonary response in the Atlanta Commuters Exposure Studies. *J. Expo Sci. Environ. Epidemiol.* **2018**, *28* (4), 337–347.

(124) Rich, D. Q.; Zhang, W.; Lin, S.; Squizzato, S.; Thurston, S. W.; van Wijngaarden, E.; Croft, D.; Masiol, M.; Hopke, P. K. Triggering of cardiovascular hospital admissions by source specific fine particle concentrations in urban centers of New York State. *Environ. Int.* **2019**, *126*, 387–394.

(125) Pun, V. C.; Tian, L. W.; Ho, K. F. Particulate matter from resuspended mineral dust and emergency cause-specific respiratory hospitalizations in Hong Kong. *Atmos. Environ.* **2017**, *165*, 191–197.

(126) Heo, J.; Schauer, J. J.; Yi, O.; Paek, D.; Kim, H.; Yi, S. M. Fine particle air pollution and mortality: importance of specific sources and chemical species. *Epidemiology* **2014**, *25* (3), 379–88.

(127) Bell, M. L.; Ebisu, K.; Leaderer, B. P.; Gent, J. F.; Lee, H. J.; Koutrakis, P.; Wang, Y.; Dominici, F.; Peng, R. D. Associations of PM_{2.5} constituents and sources with hospital admissions: analysis of four counties in Connecticut and Massachusetts (USA) for persons > / = 65 years of age. *Environ. Health Perspect* **2014**, *122* (2), 138–44.

(128) Lall, R.; Ito, K.; Thurston, G. D. Distributed lag analyses of daily hospital admissions and source-apportioned fine particle air pollution. *Environ. Health Perspect* **2011**, *119* (4), 455–60.

(129) Halonen, J. I.; Lanki, T.; Yli-Tuomi, T.; Tiittanen, P.; Kulmala, M.; Pekkanen, J. Particulate air pollution and acute cardiorespiratory hospital admissions and mortality among the elderly. *Epidemiology* **2009**, *20* (1), 143–53.

(130) Sarnat, J. A.; Marmur, A.; Klein, M.; Kim, E.; Russell, A. G.; Sarnat, S. E.; Mulholland, J. A.; Hopke, P. K.; Tolbert, P. E. Fine particle sources and cardiorespiratory morbidity: an application of chemical mass balance and factor analytical source-apportionment methods. *Environ. Health Perspect* **2008**, *116* (4), 459–66.

(131) Andersen, Z. J.; Wahlin, P.; Raaschou-Nielsen, O.; Scheike, T.; Loft, S. Ambient particle source apportionment and daily hospital admissions among children and elderly in Copenhagen. *J. Expo Sci. Environ. Epidemiol.* **2007**, *17* (7), 625–36.

(132) Dadvand, P.; Ostro, B.; Amato, F.; Figueras, F.; Minguillon, M. C.; Martinez, D.; Basagana, X.; Querol, X.; Nieuwenhuijsen, M. Particulate air pollution and preeclampsia: a source-based analysis. *Occup Environ. Med.* **2014**, *71* (8), 570–7.

(133) Riediker, M.; Devlin, R. B.; Griggs, T. R.; Herbst, M. C.; Bromberg, P. A.; Williams, R. W.; Cascio, W. E. Cardiovascular effects in patrol officers are associated with fine particulate matter from brake wear and engine emissions. *Part Fibre Toxicol* **2004**, *1* (1), 2.

(134) Chen, X. C.; Ward, T. J.; Cao, J. J.; Lee, S. C.; Lau, N. C.; Yim, S. H. L.; Ho, K. F. Source identification of personal exposure to fine particulate matter (PM_{2.5}) among adult residents of Hong Kong. *Atmos. Environ.* **2019**, 218.116999

(135) Beevers, S. D.; Kitwiroon, N.; Williams, M. L.; Kelly, F. J.; Ross Anderson, H.; Carslaw, D. C. Air pollution dispersion models for human exposure predictions in London. *J. Expo Sci. Environ. Epidemiol.* **2013**, *23* (6), 647–53.

(136) Segerström, D.; Eneroth, K.; Gidhagen, L.; Johansson, C.; Omstedt, G.; Nylen, A. E.; Forsberg, B. Health Impact of PM₁₀, PM_{2.5} and Black Carbon Exposure Due to Different Source Sectors in Stockholm, Gothenburg and Umea, Sweden. *Int. J. Environ. Res. Public Health* **2017**, *14*, (7).742

(137) Ostro, B.; Hu, J.; Goldberg, D.; Reynolds, P.; Hertz, A.; Bernstein, L.; Kleeman, M. J. Associations of mortality with long-term exposures to fine and ultrafine particles, species and sources: results from the California Teachers Study Cohort. *Environ. Health Perspect* **2015**, *123* (6), 549–56.

(138) Laurent, O.; Hu, J.; Li, L.; Kleeman, M. J.; Bartell, S. M.; Cockburn, M.; Escobedo, L.; Wu, J. A Statewide Nested Case-Control Study of Preterm Birth and Air Pollution by Source and Composition: California, 2001–2008. *Environ. Health Perspect* **2016**, *124* (9), 1479–86.

(139) Hu, J.; Zhang, H.; Chen, S. H.; Wiedinmyer, C.; Vandenberghe, F.; Ying, Q.; Kleeman, M. J. Predicting primary PM_{2.5} and PM_{0.1} trace composition for epidemiological studies in California. *Environ. Sci. Technol.* **2014**, *48* (9), 4971–9.

(140) Hoek, G. Methods for Assessing Long-Term Exposures to Outdoor Air Pollutants. *Curr. Environ. Health Rep* **2017**, *4* (4), 450–462.

(141) Ito, K.; Johnson, S.; Kheirbek, I.; Clougherty, J.; Pezeshki, G.; Ross, Z.; Eisl, H.; Matte, T. D. Intraurban Variation of Fine Particle Elemental Concentrations in New York City. *Environ. Sci. Technol.* **2016**, *50* (14), 7517–26.

- (142) Habre, R.; Girguis, M.; Urman, R.; Fruin, S.; Lurmann, F.; Shafer, M.; Gorski, P.; Franklin, M.; McConnell, R.; Avol, E.; Gilliland, F. Contribution of tailpipe and non-tailpipe traffic sources to quasi-ultrafine, fine and coarse particulate matter in southern California. *J. Air Waste Manag Assoc* **2021**, *71* (2), 209–230.
- (143) Tripathy, S.; Tunno, B. J.; Michanowicz, D. R.; Kinnee, E.; Shmool, J. L. C.; Gillooly, S.; Clougherty, J. E. Hybrid land use regression modeling for estimating spatio-temporal exposures to PM_{2.5}, BC, and metal components across a metropolitan area of complex terrain and industrial sources. *Sci. Total Environ.* **2019**, *673*, 54–63.
- (144) Tunno, B. J.; Tripathy, S.; Kinnee, E.; Michanowicz, D. R.; Shmool, J. L.; Cambal, L.; Chubb, L.; Roper, C.; Clougherty, J. E. Fine-Scale Source Apportionment Including Diesel-Related Elemental and Organic Constituents of PM_{2.5} across Downtown Pittsburgh. *Int. J. Environ. Res. Public Health* **2018**, *15*, (10).2177
- (145) Levy, J. I.; Clougherty, J. E.; Baxter, L. K.; Houseman, E. A.; Paciorek, C. J. Evaluating heterogeneity in indoor and outdoor air pollution using land-use regression and constrained factor analysis. *Res. Rep. - Health Eff. Inst.* **2010**, No. 152, 5–80. discussion 81–91.
- (146) Weichenthal, S.; Shekarrizfard, M.; Kulka, R.; Lakey, P. S. J.; Al-Rijleh, K.; Anowar, S.; Shiraiwa, M.; Hatzopoulou, M. Spatial variations in the estimated production of reactive oxygen species in the epithelial lung lining fluid by iron and copper in fine particulate air pollution. *Environ. Epidemiol* **2018**, *2* (3), e020.
- (147) Zhang, J. J. Y.; Sun, L.; Barrett, O.; Bertazzon, S.; Underwood, F. E.; Johnson, M. Development of land-use regression models for metals associated with airborne particulate matter in a North American city. *Atmos. Environ.* **2015**, *106*, 165–177.
- (148) Zhang, K.; Larson, T. V.; Gassett, A.; Szpiro, A. A.; Daviglius, M.; Burke, G. L.; Kaufman, J. D.; Adar, S. D. Characterizing spatial patterns of airborne coarse particulate (PM_{10–2.5}) mass and chemical components in three cities: the multi-ethnic study of atherosclerosis. *Environ. Health Perspect* **2014**, *122* (8), 823–30.
- (149) Chen, J.; de Hoogh, K.; Gulliver, J.; Hoffmann, B.; Hertel, O.; Ketzel, M.; Weinmayr, G.; Bauwelinck, M.; van Donkelaar, A.; Hvidtfeldt, U. A.; Atkinson, R.; Janssen, N. A. H.; Martin, R. V.; Samoli, E.; Andersen, Z. J.; Oftedal, B. M.; Stafoggia, M.; Bellander, T.; Strak, M.; Wolf, K.; Vienneau, D.; Brunekreef, B.; Hoek, G. Development of Europe-Wide Models for Particle Elemental Composition Using Supervised Linear Regression and Random Forest. *Environ. Sci. Technol.* **2020**, *54* (24), 15698–15709.
- (150) de Hoogh, K.; Wang, M.; Adam, M.; Badaloni, C.; Beelen, R.; Birk, M.; Cesaroni, G.; Cirach, M.; Declercq, C.; Dedele, A.; Dons, E.; de Nazelle, A.; Eeftens, M.; Eriksen, K.; Eriksson, C.; Fischer, P.; Grazuleviciene, R.; Gryparis, A.; Hoffmann, B.; Jerrett, M.; Katsouyanni, K.; Iakovides, M.; Lanki, T.; Lindley, S.; Madsen, C.; Molter, A.; Mosler, G.; Nador, G.; Nieuwenhuijsen, M.; Pershagen, G.; Peters, A.; Phuleria, H.; Probst-Hensch, N.; Raaschou-Nielsen, O.; Quass, U.; Ranzi, A.; Stephanou, E.; Sugiri, D.; Schwarze, P.; Tsai, M. Y.; Yli-Tuomi, T.; Varro, M. J.; Vienneau, D.; Weinmayr, G.; Brunekreef, B.; Hoek, G. Development of land use regression models for particle composition in twenty study areas in Europe. *Environ. Sci. Technol.* **2013**, *47* (11), 5778–86.
- (151) Dirgawati, M.; Heyworth, J. S.; Wheeler, A. J.; McCaul, K. A.; Blake, D.; Boeyen, J.; Cope, M.; Yeap, B. B.; Nieuwenhuijsen, M.; Brunekreef, B.; Hinwood, A. Development of Land Use Regression models for particulate matter and associated components in a low air pollutant concentration airshed. *Atmos. Environ.* **2016**, *144*, 69–78.
- (152) Huang, C. S.; Liao, H. T.; Lin, T. H.; Chang, J. C.; Lee, C. L.; Yip, E. C. W.; Wu, Y. L.; Wu, C. F. Evaluation of Using Satellite-Derived Aerosol Optical Depth in Land Use Regression Models for Fine Particulate Matter and Its Elemental Composition. *Atmosphere* **2021**, *12*, (8).1018
- (153) Wu, C. F.; Lin, H. I.; Ho, C. C.; Yang, T. H.; Chen, C. C.; Chan, C. C. Modeling horizontal and vertical variation in intraurban exposure to PM_{2.5} concentrations and compositions. *Environ. Res.* **2014**, *133*, 96–102.
- (154) Eeftens, M.; Beelen, R.; de Hoogh, K.; Bellander, T.; Cesaroni, G.; Cirach, M.; Declercq, C.; Dedele, A.; Dons, E.; de Nazelle, A.; Dimakopoulou, K.; Eriksen, K.; Falq, G.; Fischer, P.; Galassi, C.; Grazuleviciene, R.; Heinrich, J.; Hoffmann, B.; Jerrett, M.; Keidel, D.; Korek, M.; Lanki, T.; Lindley, S.; Madsen, C.; Molter, A.; Nador, G.; Nieuwenhuijsen, M.; Nonnemacher, M.; Pedeli, X.; Raaschou-Nielsen, O.; Patelarou, E.; Quass, U.; Ranzi, A.; Schindler, C.; Stempfelet, M.; Stephanou, E.; Sugiri, D.; Tsai, M. Y.; Yli-Tuomi, T.; Varro, M. J.; Vienneau, D.; Klot, S.; Wolf, K.; Brunekreef, B.; Hoek, G. Development of Land Use Regression models for PM(2.5), PM(2.5) absorbance, PM(10) and PM(coarse) in 20 European study areas; results of the ESCAPE project. *Environ. Sci. Technol.* **2012**, *46* (20), 11195–205.
- (155) Beelen, R.; Hoek, G.; Vienneau, D.; Eeftens, M.; Dimakopoulou, K.; Pedeli, X.; Tsai, M. Y.; Kunzli, N.; Schikowski, T.; Marcon, A.; Eriksen, K. T.; Raaschou-Nielsen, O.; Stephanou, E.; Patelarou, E.; Lanki, T.; Yli-Tuomi, T.; Declercq, C.; Falq, G.; Stempfelet, M.; Birk, M.; Cyrus, J.; von Klot, S.; Nador, G.; Varro, M. J.; Dedele, A.; Grazuleviciene, R.; Molter, A.; Lindley, S.; Madsen, C.; Cesaroni, G.; Ranzi, A.; Badaloni, C.; Hoffmann, B.; Nonnemacher, M.; Kraemer, U.; Kuhlbusch, T.; Cirach, M.; de Nazelle, A.; Nieuwenhuijsen, M.; Bellander, T.; Korek, M.; Olsson, D.; Stromgren, M.; Dons, E.; Jerrett, M.; Fischer, P.; Wang, M.; Brunekreef, B.; de Hoogh, K. Development of NO₂ and NO_x land use regression models for estimating air pollution exposure in 36 study areas in Europe - The ESCAPE project. *Atmos. Environ.* **2013**, *72*, 10–23.
- (156) Brokamp, C.; Jandarov, R.; Rao, M. B.; LeMasters, G.; Ryan, P. Exposure assessment models for elemental components of particulate matter in an urban environment: A comparison of regression and random forest approaches. *Atmos. Environ.* **2017**, *151*, 1–11.
- (157) Stanek, L. W.; Sacks, J. D.; Dutton, S. J.; Dubois, J.-J. B. Attributing health effects to apportioned components and sources of particulate matter: An evaluation of collective results. *Atmos. Environ.* **2011**, *45* (32), 5655–5663.
- (158) Atkinson, R. W.; Analitis, A.; Samoli, E.; Fuller, G. W.; Green, D. C.; Mudway, I. S.; Anderson, H. R.; Kelly, F. J. Short-term exposure to traffic-related air pollution and daily mortality in London, UK. *J. Expo Sci. Environ. Epidemiol* **2016**, *26* (2), 125–32.
- (159) Yang, Y.; Ruan, Z.; Wang, X.; Yang, Y.; Mason, T. G.; Lin, H.; Tian, L. Short-term and long-term exposures to fine particulate matter constituents and health: A systematic review and meta-analysis. *Environ. Pollut.* **2019**, *247*, 874–882.
- (160) EPA, U. S. *Integrated Science Assessment (ISA) for Particulate Matter*; U.S. Environmental Protection Agency: Washington, DC, 2019.
- (161) Lavigne, A.; Freni Sterrantino, A.; Liverani, S.; Blangiardo, M.; de Hoogh, K.; Molitor, J.; Hansell, A. Associations between metal constituents of ambient particulate matter and mortality in England: an ecological study. *BMJ. Open* **2019**, *9* (12), e030140.
- (162) Thurston, G. D.; Burnett, R. T.; Turner, M. C.; Shi, Y.; Krewski, D.; Lall, R.; Ito, K.; Jerrett, M.; Gapstur, S. M.; Diver, W. R.; Pope, C. A. Ischemic Heart Disease Mortality and Long-Term Exposure to Source-Related Components of U.S. Fine Particle Air Pollution. *Environ. Health Perspect* **2016**, *124* (6), 785–94.
- (163) Weichenthal, S.; Lavigne, E.; Traub, A.; Umbrio, D.; You, H.; Pollitt, K.; Shin, T.; Kulka, R.; Stieb, D. M.; Korsiak, J.; Jessiman, B.; Brook, J. R.; Hatzopoulou, M.; Evans, G.; Burnett, R. T. Association of Sulfur, Transition Metals, and the Oxidative Potential of Outdoor PM_{2.5} with Acute Cardiovascular Events: A Case-Crossover Study of Canadian Adults. *Environ. Health Perspect.* **2021**, *129*, (10). DOI: 10.1289/EHP9449
- (164) Ostro, B.; Tobias, A.; Querol, X.; Alastuey, A.; Amato, F.; Pey, J.; Perez, N.; Sunyer, J. The effects of particulate matter sources on daily mortality: a case-crossover study of Barcelona, Spain. *Environ. Health Perspect* **2011**, *119* (12), 1781–7.
- (165) Zhou, L.; Tao, Y.; Li, H.; Niu, Y.; Li, L.; Kan, H.; Xie, J.; Chen, R. Acute effects of fine particulate matter constituents on

- cardiopulmonary function in a panel of COPD patients. *Sci. Total Environ.* **2021**, *770*, 144753.
- (166) Golan, R.; Ladva, C.; Greenwald, R.; Krall, J. R.; Raysoni, A. U.; Kewada, P.; Winqvist, A.; Flanders, W. D.; Liang, D.; Sarnat, J. A. Acute pulmonary and inflammatory response in young adults following a scripted car commute. *Air Quality, Atmosphere & Health* **2018**, *11* (2), 123–136.
- (167) Chen, J.; Rodopoulou, S.; de Hoogh, K.; Strak, M.; Andersen, Z. J.; Atkinson, R.; Bauwelinck, M.; Bellander, T.; Brandt, J.; Cesaroni, G.; Concina, H.; Fecht, D.; Forastiere, F.; Gulliver, J.; Hertel, O.; Hoffmann, B.; Hvidtfeldt, U. A.; Janssen, N. A. H.; Jöckel, K. H.; Jørgensen, J.; Katsouyanni, K.; Ketzel, M.; Klompaker, J. O.; Lager, A.; Leander, K.; Liu, S.; Ljungman, P.; MacDonald, C. J.; Magnusson, P. K. E.; Mehta, A.; Nagel, G.; Oftedal, B.; Pershagen, G.; Peters, A.; Raaschou-Nielsen, O.; Renzi, M.; Rizzuto, D.; Samoli, E.; van der Schouw, Y. T.; Schramm, S.; Schwarze, P.; Sigsgaard, T.; Sørensen, M.; Stafoggia, M.; Tjønneland, A.; Vienneau, D.; Weinmayr, G.; Wolf, K.; Brunekreef, B.; Hoek, G. Long-Term Exposure to Fine Particle Elemental Components and Natural and Cause-Specific Mortality—a Pooled Analysis of Eight European Cohorts within the ELAPSE Project. *Environ. Health Perspect* **2021**, *129* (4), 47009.
- (168) Zhang, Z.; Weichenthal, S.; Kwong, J. C.; Burnett, R. T.; Hatzopoulou, M.; Jerrett, M.; Donkelaar, A. V.; Bai, L.; Martin, R. V.; Copes, R.; Lu, H.; Lakey, P.; Shiraiwa, M.; Chen, H. Long-term exposure to iron and copper in fine particulate air pollution and their combined impact on reactive oxygen species concentration in lung fluid: a population-based cohort study of cardiovascular disease incidence and mortality in Toronto, Canada. *Int. J. Epidemiol* **2021**, *50* (2), 589–601.
- (169) Fuertes, E.; MacIntyre, E.; Agius, R.; Beelen, R.; Brunekreef, B.; Bucci, S.; Cesaroni, G.; Cirach, M.; Cyrys, J.; Forastiere, F.; Gehring, U.; Gruziera, O.; Hoffmann, B.; Jedynska, A.; Keuken, M.; Klümper, C.; Kooter, I.; Korek, M.; Krämer, U.; Mölter, A.; Nieuwenhuijsen, M.; Pershagen, G.; Porta, D.; Postma, D. S.; Simpson, A.; Smit, H. A.; Sugiri, D.; Sunyer, J.; Wang, M.; Heinrich, J. Associations between particulate matter elements and early-life pneumonia in seven birth cohorts: results from the ESCAPE and TRANSPHORM projects. *Int. J. Hyg Environ. Health* **2014**, *217* (8), 819–29.
- (170) Gehring, U.; Beelen, R.; Eeftens, M.; Hoek, G.; de Hoogh, K.; de Jongste, J. C.; Keuken, M.; Koppelman, G. H.; Meliefste, K.; Oldenwening, M.; Postma, D. S.; van Rossem, L.; Wang, M.; Smit, H. A.; Brunekreef, B. Particulate matter composition and respiratory health: the PIAMA Birth Cohort study. *Epidemiology* **2015**, *26* (3), 300–9.
- (171) Sun, X.; Luo, X.; Zhao, C.; Zhang, B.; Tao, J.; Yang, Z.; Ma, W.; Liu, T. The associations between birth weight and exposure to fine particulate matter (PM_{2.5}) and its chemical constituents during pregnancy: A meta-analysis. *Environ. Pollut.* **2016**, *211*, 38–47.
- (172) Smith, R. B.; Beevers, S. D.; Gulliver, J.; Dajnak, D.; Fecht, D.; Blangiardo, M.; Douglass, M.; Hansell, A. L.; Anderson, H. R.; Kelly, F. J.; Toledano, M. B. Impacts of air pollution and noise on risk of preterm birth and stillbirth in London. *Environ. Int.* **2020**, *134*, 105290.
- (173) Lubczyńska, M. J.; Muetzel, R. L.; El Marroun, H.; Basagaña, X.; Strak, M.; Denault, W.; Jaddoe, V. W. V.; Hillegers, M.; Vernooij, M. W.; Hoek, G.; White, T.; Brunekreef, B.; Tiemeier, H.; Guxens, M. Exposure to Air Pollution during Pregnancy and Childhood, and White Matter Microstructure in Preadolescents. *Environ. Health Perspect* **2020**, *128* (2), 27005.
- (174) von Ehrenstein, O. S.; Aralis, H.; Cockburn, M.; Ritz, B. In Utero Exposure to Toxic Air Pollutants and Risk of Childhood Autism. *Epidemiology* **2014**, *25* (6), 851–858.
- (175) Mudway, I. S.; Kelly, F. J.; Holgate, S. T. Oxidative stress in air pollution research. *Free Radical Biol. Med.* **2020**, *151*, 2–6.
- (176) Gray, D. L.; Wallace, L. A.; Brinkman, M. C.; Buehler, S. S.; La Londe, C. Respiratory and cardiovascular effects of metals in ambient particulate matter: a critical review. *Reviews of environmental contamination and toxicology* **2015**, *234*, 135–203.
- (177) Wu, W.; Bromberg, P. A.; Samet, J. M. Zinc ions as effectors of environmental oxidative lung injury. *Free Radic Biol. Med.* **2013**, *65*, 57–69.
- (178) Holme, J. A.; Brinchmann, B. C.; Refsnes, M.; Lag, M.; Ovrevik, J. Potential role of polycyclic aromatic hydrocarbons as mediators of cardiovascular effects from combustion particles. *Environ. Health* **2019**, *18* (1), 74.
- (179) Sharma, J.; Parsai, K.; Raghuvanshi, P.; Ali, S. A.; Tiwari, V.; Bhargava, A.; Mishra, P. K. Emerging role of mitochondria in airborne particulate matter-induced immunotoxicity. *Environ. Pollut.* **2021**, *270*, 116242.
- (180) Guastadisegni, C.; Kelly, F. J.; Cassee, F. R.; Gerlofs-Nijland, M. E.; Janssen, N. A.; Pozzi, R.; Brunekreef, B.; Sandström, T.; Mudway, I. Determinants of the proinflammatory action of ambient particulate matter in immortalized murine macrophages. *Environ. Health Perspect* **2010**, *118* (12), 1728–34.
- (181) Selley, L.; Schuster, L.; Marbach, H.; Forsthuber, T.; Forbes, B.; Gant, T. W.; Sandstrom, T.; Camina, N.; Athersuch, T. J.; Mudway, I.; Kumar, A. Brake dust exposure exacerbates inflammation and transiently compromises phagocytosis in macrophages. *Metalomics* **2019**. DOI: 10.1039/C9MT00253G
- (182) Xu, M.; Li, F.; Wang, M.; Zhang, H.; Xu, L.; Adcock, I. M.; Chung, K. F.; Zhang, Y. Protective effects of VGX-1027 in PM_{2.5}-induced airway inflammation and bronchial hyperresponsiveness. *Eur. J. Pharmacol.* **2019**, *842*, 373–383.
- (183) Kim, E.-A.; Koh, B. Utilization of road dust chemical profiles for source identification and human health impact assessment. *Sci. Rep.* **2020**, *10*, (1).
- (184) Gerlofs-Nijland, M. E.; Dormans, J. A.; Bloemen, H. J.; Leseman, D. L.; John, A.; Boere, F.; Kelly, F. J.; Mudway, I. S.; Jimenez, A. A.; Donaldson, K.; Guastadisegni, C.; Janssen, N. A.; Brunekreef, B.; Sandstrom, T.; van Bree, L.; Cassee, F. R. Toxicity of coarse and fine particulate matter from sites with contrasting traffic profiles. *Inhal Toxicol* **2007**, *19* (13), 1055–69.
- (185) Happonen, M. S.; Hirvonen, M. R.; Hälinen, A. I.; Jalava, P. I.; Pennanen, A. S.; Sillanpää, M.; Hillamo, R.; Salonen, R. O. Seasonal variation in chemical composition of size-segregated urban air particles and the inflammatory activity in the mouse lung. *Inhal Toxicol* **2010**, *22* (1), 17–32.
- (186) Kreider, M. L.; Doyle-Eisele, M.; Russell, R. G.; McDonald, J. D.; Panko, J. M. Evaluation of potential for toxicity from subacute inhalation of tire and road wear particles in rats. *Inhal Toxicol* **2012**, *24* (13), 907–17.
- (187) Kreider, M. L.; Unice, K. M.; Panko, J. M. Human health risk assessment of Tire and Road Wear Particles (TRWP) in air. *Human and Ecological Risk Assessment: An International Journal* **2020**, *26* (10), 2567–2585.
- (188) Gasser, M.; Riediker, M.; Mueller, L.; Perrenoud, A.; Blank, F.; Gehr, P.; Rothen-Rutishauser, B. Toxic effects of brake wear particles on epithelial lung cells in vitro. *Part Fibre Toxicol* **2009**, *6*, 30.
- (189) Figliuzzi, M.; Tironi, M.; Longaretti, L.; Mancini, A.; Teoldi, F.; Sangalli, F.; Remuzzi, A. Copper-dependent biological effects of particulate matter produced by brake systems on lung alveolar cells. *Arch. Toxicol.* **2020**, *94* (9), 2965–2979.
- (190) Barosova, H.; Chortarea, S.; Peikertova, P.; Clift, M. J. D.; Petri-Fink, A.; Kukutschova, J.; Rothen-Rutishauser, B. Biological response of an in vitro human 3D lung cell model exposed to brake wear debris varies based on brake pad formulation. *Arch. Toxicol.* **2018**, *92* (7), 2339–2351.
- (191) Poland, C. A.; Duffin, R. The toxicology of chrysotile-containing brake debris: implications for mesothelioma. *Crit Rev. Toxicol* **2019**, *49* (1), 11–35.
- (192) Gerlofs-Nijland, M. E.; Bokkers, B. G. H.; Sachse, H.; Reijnders, J. J. E.; Gustafsson, M.; Boere, A. J. F.; Fokkens, P. F. H.; Leseman, D.; Augsburg, K.; Cassee, F. R. Inhalation toxicity profiles of particulate matter: a comparison between brake wear with other sources of emission. *Inhal Toxicol* **2019**, 1–10.
- (193) Lindbom, J.; Gustafsson, M.; Blomqvist, G.; Dahl, A.; Gudmundsson, A.; Swietlicki, E.; Ljungman, A. G. Exposure to wear

- particles generated from studded tires and pavement induces inflammatory cytokine release from human macrophages. *Chem. Res. Toxicol.* **2006**, *19* (4), 521–30.
- (194) Karlsson, H. L.; Ljungman, A. G.; Lindbom, J.; Möller, L. Comparison of genotoxic and inflammatory effects of particles generated by wood combustion, a road simulator and collected from street and subway. *Toxicol. Lett.* **2006**, *165* (3), 203–11.
- (195) Karlsson, H. L.; Holgersson, A.; Möller, L. Mechanisms related to the genotoxicity of particles in the subway and from other sources. *Chem. Res. Toxicol.* **2008**, *21* (3), 726–31.
- (196) Lindbom, J.; Gustafsson, M.; Blomqvist, G.; Dahl, A.; Gudmundsson, A.; Swietlicki, E.; Ljungman, A. G. Wear particles generated from studded tires and pavement induces inflammatory reactions in mouse macrophage cells. *Chem. Res. Toxicol.* **2007**, *20* (6), 937–46.
- (197) Karlsson, H.; Lindbom, J.; Ghafouri, B.; Lindahl, M.; Tagesson, C.; Gustafsson, M.; Ljungman, A. G. Wear particles from studded tires and granite pavement induce pro-inflammatory alterations in human monocyte-derived macrophages: a proteomic study. *Chem. Res. Toxicol.* **2011**, *24* (1), 45–53.
- (198) Gualtieri, M.; Rigamonti, L.; Galeotti, V.; Camatini, M. Toxicity of tire debris extracts on human lung cell line A549. *Toxicology in Vitro* **2005**, *19* (7), 1001–1008.
- (199) Gualtieri, M.; Mantecca, P.; Cetta, F.; Camatini, M. Organic compounds in tire particle induce reactive oxygen species and heat-shock proteins in the human alveolar cell line A549. *Environ. Int.* **2008**, *34* (4), 437–42.
- (200) Beretta, E.; Gualtieri, M.; Botto, L.; Palestini, P.; Miserocchi, G.; Camatini, M. Organic extract of tire debris causes localized damage in the plasma membrane of human lung epithelial cells. *Toxicol. Lett.* **2007**, *173* (3), 191–200.
- (201) Gottipolu, R. R.; Landa, E. R.; Schladweiler, M. C.; McGee, J. K.; Ledbetter, A. D.; Richards, J. H.; Wallenborn, G. J.; Kodavanti, U. P. Cardiopulmonary responses of intratracheally instilled tire particles and constituent metal components. *Inhal Toxicol* **2008**, *20* (5), 473–84.
- (202) Mantecca, P.; Sancini, G.; Moschini, E.; Farina, F.; Gualtieri, M.; Rohr, A.; Miserocchi, G.; Palestini, P.; Camatini, M. Lung toxicity induced by intratracheal instillation of size-fractionated tire particles. *Toxicol. Lett.* **2009**, *189* (3), 206–14.
- (203) Mantecca, P.; Farina, F.; Moschini, E.; Gallinotti, D.; Gualtieri, M.; Rohr, A.; Sancini, G.; Palestini, P.; Camatini, M. Comparative acute lung inflammation induced by atmospheric PM and size-fractionated tire particles. *Toxicol. Lett.* **2010**, *198* (2), 244–54.
- (204) Miller, M. R.; Raftis, J. B.; Langrish, J. P.; McLean, S. G.; Samutritai, P.; Connell, S. P.; Wilson, S.; Vesey, A. T.; Fokkens, P. H. B.; Boere, A. J. F.; Krystek, P.; Campbell, C. J.; Hadoke, P. W. F.; Donaldson, K.; Cassee, F. R.; Newby, D. E.; Duffin, R.; Mills, N. L. Inhaled Nanoparticles Accumulate at Sites of Vascular Disease. *ACS Nano* **2017**, *11* (5), 4542–4552.
- (205) Maher, B. A.; Ahmed, I. A.; Karloukovski, V.; MacLaren, D. A.; Foulds, P. G.; Allsop, D.; Mann, D. M.; Torres-Jardon, R.; Calderon-Garciduenas, L. Magnetite pollution nanoparticles in the human brain. *Proc. Natl. Acad. Sci. U. S. A.* **2016**, *113* (39), 10797–801.
- (206) Maher, B. A.; González-Maciel, A.; Reynoso-Robles, R.; Torres-Jardón, R.; Calderón-Garcidueñas, L. Iron-rich air pollution nanoparticles: An unrecognized environmental risk factor for myocardial mitochondrial dysfunction and cardiac oxidative stress. *Environ. Res.* **2020**, *188*, 109816.
- (207) Gustafsson, A.; Kraus, A. M.; Gorzszak, A.; Lundh, T.; Gerde, P. Isolation and characterization of a respirable particle fraction from residential house-dust. *Environ. Res.* **2018**, *161*, 284–290.
- (208) Beddows, D. C. S.; Harrison, R. M. PM10 and PM2.5 emission factors for non-exhaust particles from road vehicles: Dependence upon vehicle mass and implications for battery electric vehicles. *Atmos. Environ.* **2021**, *244*, 117886.
- (209) Timmers, V. R. J. H.; Achten, P. A. J. Non-exhaust PM emissions from electric vehicles. *Atmos. Environ.* **2016**, *134*, 10–17.
- (210) DfT. *Road Traffic Forecasts 2018: Moving Britain Ahead*; DfT: London, 2018.
- (211) NAEI In *Road Transport Emission Factor*, 2018.
- (212) Hooftman, N.; Oliveira, L.; Messagie, M.; Coosemans, T.; Van Mierlo, J. Environmental Analysis of Petrol, Diesel and Electric Passenger Cars in a Belgian Urban Setting. *Energies* **2016**, *9* (2), 84.
- (213) De Coensel, B.; Can, A.; Degraeuwe, B.; De Vlieger, I.; Botteldooren, D. Effects of traffic signal coordination on noise and air pollutant emissions. *Environ. Modell Softw* **2012**, *35*, 74–83.
- (214) Vieira, T.; Sandberg, U.; Erlingsson, S. Negative texture, positive for the environment: effects of horizontal grinding of asphalt pavements. *Road Materials and Pavement Design* **2021**, *22* (1), 1–22.
- (215) Egede, P. *Environmental Assessment of Lightweight Electric Vehicles*; Springer International Publishing: Basel, Switzerland, 2017.
- (216) Shaffer, B.; Auffhammer, M.; Samaras, C. *Make Electric Vehicles Lighter to Maximize Climate and Safety Benefits*. In Nature Publishing Group, 2021.
- (217) Hesse, D.; Hamatschek, C.; Augsburg, K.; Weigelt, T.; Prahst, A.; Gramstat, S. Testing of Alternative Disc Brakes and Friction Materials Regarding Brake Wear Particle Emissions and Temperature Behavior. *Atmosphere* **2021**, *12* (4), 436.
- (218) Perricone, G.; Matějka, V.; Alemani, M.; Valota, G.; Bonfanti, A.; Ciotti, A.; Olofsson, U.; Söderberg, A.; Wahlström, J.; Nosko, O.; Straffellini, G.; Gialanella, S.; Ibrahim, M. A concept for reducing PM10 emissions for car brakes by 50%. *Wear* **2018**, *396–397*, 135–145.
- (219) Fieldhouse, J. D.; Gelb, J. New Developments of an On-Vehicle Brake Pad Waste Collection System. *SAE International Journal of Passenger Cars - Mechanical Systems* **2016**, *9* (3), 1245–1254.
- (220) MANN+HUMMEL Brake dust Particle Filter. <https://oem.mann-hummel.com/en/oem-products/fine-dust-filters/brake-dust-particle-filter.html> (accessed 2021-11-20).
- (221) Vojtišek-Lom, M.; Vaculík, M.; Pechout, M.; Hopan, F.; Arul Raj, A. F.; Penumarti, S.; Horák, J. S.; Popovicheva, O.; Ondráček, J.; Doušová, B. Effects of braking conditions on nanoparticle emissions from passenger car friction brakes. *Science of The Total Environment* **2021**, *788*, 147779.
- (222) ETRMA Addressing Tyre and Road Wear Particles. <https://www.tyreandroadwear.com/> (accessed 2021-11-20).
- (223) CORDIS Low particle Emissions and Low Noise Tyres. <https://cordis.europa.eu/project/id/955387> (accessed 2021-11-20).
- (224) Tyre-Collective Tyres: The stealthy pollutant you never thought about. <https://www.thetyrecollective.com/> (accessed 2021-11-20).
- (225) Project-uCARE Project uCARE (accessed 2021-11-20).
- (226) Modales Adapting driver behaviour for lower emissions. <https://modales-project.eu/> (accessed 2021-11-20).
- (227) Jacobson, T. *Belägningslitaget från dubbade fordon: beräkning av det årliga dubbslitaget 1996–1999*; Swedish National Road and Transport Research Institute: Linköping, Sweden, 1999.
- (228) Norman, M.; Sundvor, I.; Denby, B. R.; Johansson, C.; Gustafsson, M.; Blomqvist, G.; Janhäll, S. Modelling road dust emission abatement measures using the NORTRIP model: Vehicle speed and studded tyre reduction. *Atmos. Environ.* **2016**, *134*, 96–108.
- (229) Berge, T.; Storeheier, S. Å. *Low Noise Pavements in a Nordic Climate. Results from a Four Year Project in Norway* **2009**, *2009*, 359–367.
- (230) Ejsmont, J.; Sandberg, U.; Świczko-Żurek, B.; Mioduszewski, P. Tyre/road noise reduction by a poroelastic road surface. In *43rd International Congress on Noise Control Engineering: Improving the World Through Noise Control, INTERNOISE 2014, 16 November 2014 through 19 November 2014*, 2014.
- (231) Peng, B.; Han, S.; Han, X.; Zhang, H. Laboratory and field evaluation of noise characteristics of porous asphalt pavement. *Int. J. Pavement Eng.* **2021**, *1*.
- (232) DIN EN 15429-3 Sweepers - Part 3: Efficiency of particulate matter collection - Testing and Evaluation. <https://www.en-standard>.

eu/din-en-15429-3-sweepers-part-3-efficiency-of-particulate-matter-collection-testing-and-evaluation/.

(233) Amato, F.; Querol, X.; Alastuey, A.; Pandolfi, M.; Moreno, T.; Gracia, J.; Rodriguez, P. Evaluating urban PM10 pollution benefit induced by street cleaning activities. *Atmos. Environ.* **2009**, *43* (29), 4472–4480.

(234) Bogacki, M.; Oleniacz, R.; Rzeszutek, M.; Szulecka, A.; Mazur, M. In *The impact of street cleaning on particulate matter air concentrations: A case study of a street canyon in Krakow (Poland)*; Dziopak, J., Slys, D., Eds.; EDP Sciences, 2018.

(235) Karanasiou, A.; Amato, F.; Moreno, T.; Lumbreras, J.; Borge, R.; Linares, C.; Boldo, E.; Alastuey, A.; Querol, X. Road dust emission sources and assessment of street washing effect. *Aerosol Air Qual. Res.* **2014**, *14* (3), 734–743.

(236) Kryłów, M.; Generowicz, A. Impact of street sweeping and washing on the pm10 and PM2.5 concentrations in cracow (Poland). *Rocz. Ochr. R.* **2019**, *21* (1), 691–711.

(237) Järskog, I.; Blomqvist, G.; Gustafsson, M.; Janhäll, S. *Utvärdering av städmaskiners förmåga att reducera vägdammförrådet i gatu- och tunnelsemiljöer: En fältstudie i Trondheim 2016*; 03476030 (ISSN); Statens väg- och transportforskningsinstitut: Linköping, 2017, 2017; p 39.

(238) Janhäll, S.; Gustafsson, M.; Andersson, K.; Järskog, I.; Lindström, T. *Utvärdering av städmaskiners förmåga att reducera vägdammförrådet i gatu- och tunnelsemiljöer i Trondheim*; 03476030 (ISSN); Statens väg- och transportforskningsinstitut: Linköping, 2016, 2016; p 30.

(239) Gulia, S.; Goyal, P.; Goyal, S. K.; Kumar, R. Re-suspension of road dust: contribution, assessment and control through dust suppressants—a review. *International Journal of Environmental Science and Technology* **2019**, *16* (3), 1717–1728.

(240) Lundberg, J.; Gustafsson, M.; Janhäll, S.; Eriksson, O.; Blomqvist, G.; Erlingsson, S. Temporal Variation of Road Dust Load and Its Size Distribution—A Comparative Study of a Porous and a Dense Pavement. *Water, Air, Soil Pollut.* **2020**, *231*, (12). DOI: 10.1007/s11270-020-04923-1

(241) Norman, M.; Johansson, C. In *Emission of PM10 and coarse particles from “silent” asphalt*, 22nd International Transport and Air Pollution Conference, Zurich, Switzerland, 2017; Zurich, Switzerland, 2017.

(242) UNECE Particle Measurement Programme (PMP). <https://wiki.unece.org/pages/viewpage.action?pageId=2523173>.

(243) Martins, M.; Lawrence, J.; Ferguson, S.; Wolfson, J. M.; Koutrakis, P. Development and evaluation of a mobile laboratory for collecting short-duration near-road fine and coarse ambient particle and road dust samples. *J. Air Waste Manage. Assoc.* **2021**, *71* (2), 118–128.

(244) Lundberg, J.; Blomqvist, G.; Gustafsson, M.; Janhäll, S.; Järskog, I. Wet Dust Sampler—a Sampling Method for Road Dust Quantification and Analyses. *Water, Air, Soil Pollut.* **2019**, *230*, (8). DOI: 10.1007/s11270-019-4226-6

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

(246) Mathissen, M.; Scheer, V.; Kirchner, U.; Vogt, R.; Benter, T. Non-exhaust PM emission measurements of a light duty vehicle with a mobile trailer. *Atmos. Environ.* **2012**, *59*, 232–242.

(247) Strak, M.; Janssen, N. A.; Godri, K. J.; Gosens, I.; Mudway, I. S.; Cassee, F. R.; Lebret, E.; Kelly, F. J.; Harrison, R. M.; Brunekreef, B.; Steenhof, M.; Hoek, G. Respiratory Health Effects of Airborne Particulate Matter: The Role of Particle Size, Composition, and Oxidative Potential—The RAPTES Project. *Environ. Health Perspect* **2012**, *120* (8), 1183–1189.

(248) Steenhof, M.; Mudway, I. S.; Gosens, I.; Hoek, G.; Godri, K. J.; Kelly, F. J.; Harrison, R. M.; Pieters, R. H.; Cassee, F. R.; Lebret, E.; Brunekreef, B. A.; Strak, M.; Janssen, N. A. Acute nasal pro-inflammatory response to air pollution depends on characteristics

other than particle mass concentration or oxidative potential: the RAPTES project. *Occup Environ. Med.* **2013**, *70* (5), 341–8.

(249) Kunzli, N.; Mudway, I. S.; Gotschi, T.; Shi, T.; Kelly, F. J.; Cook, S.; Burney, P.; Forsberg, B.; Gauderman, J. W.; Hazenkamp, M. E.; Heinrich, J.; Jarvis, D.; Norback, D.; Payo-Losa, F.; Poli, A.; Sunyer, J.; Borm, P. J. Comparison of oxidative properties, light absorbance, total and elemental mass concentration of ambient PM2.5 collected at 20 European sites. *Environ. Health Perspect* **2006**, *114* (5), 684–90.

(250) Antonini, J. M.; Roberts, J. R. Chromium in stainless steel welding fume suppresses lung defense responses against bacterial infection in rats. *J. Immunotoxicol* **2007**, *4* (2), 117–27.

(251) Palmer, K.; Coggon, D. Does occupational exposure to iron promote infection? *Occupational and Environmental Medicine* **1997**, *54*, 529–534.

(252) Nelson, R. J. Dietary iron and colorectal cancer. *Free rad Biol. Med.* **1992**, *12*, 161–168.

(253) Lopez, C. A.; Skaar, E. P. The Impact of Dietary Transition Metals on Host-Bacterial Interactions. *Cell Host & Microbe* **2018**, *23* (6), 737–748.

(254) Vedal, S.; Campen, M. J.; McDonald, J. D.; Larson, T. V.; Sampson, P. D.; Sheppard, L.; Simpson, C. D.; Szpiro, A. A. National Particle Component Toxicity (NPACT) initiative report on cardiovascular effects. *Res. Rep. - Health Eff. Inst.* **2013**, No. 178, 5–8.

(255) WHO Review of evidence on health aspects of air pollution—REVIHAAP project, technical report. http://www.euro.who.int/_data/assets/pdf_file/0004/193108/REVIHAAP-Final-technical-report-final-version.pdf?ua=1 (accessed 2021-3-15).

(256) Kelly, F.; Anderson, H. R.; Armstrong, B.; Atkinson, R.; Barratt, B.; Beevers, S.; Derwent, D.; Green, D.; Mudway, I.; Wilkinson, P. The impact of the congestion charging scheme on air quality in London. Part 2. Analysis of the oxidative potential of particulate matter. *Res. Rep. - Health Eff. Inst.* **2011**, No. 155, 73–144.

(257) Adams, P. C. Epidemiology and diagnostic testing for hemochromatosis and iron overload. *International journal of laboratory hematology* **2015**, *37* (Suppl 1), 25–30.