4. Policy responses to tackle particulate matter from non-exhaust emissions

This chapter identifies currently available technological solutions to reduce non-exhaust PM emissions from wear and road dust resuspension and reviews policy measures that have an effect on non-exhaust emissions. The chapter then develops a framework for the design of an efficient public policy instrument to address non-exhaust emissions. Finally, it provides an overview of the main uncertainties and data gaps on non-exhaust emissions and proposes next steps for moving forward in better understanding and mitigating PM emissions from non-exhaust sources. Given the high proportion of non-exhaust emissions generated by tyre wear, priority should be placed on measures that seek to reduce PM emissions from this source in particular, namely vehicle lightweighting and regulations on tyre composition. Priority should also be placed on conducting research to better understand the mitigation potential of different policy options.

Few policy measures currently target non-exhaust emissions. Two broad approaches can be taken to reduce non-exhaust emissions from road traffic, namely either lowering emission factors or reducing vehicle-kilometres travelled.[1] Exceptions come in the form of regulations regarding brake and tyre material, and mitigating measures such as street washing that are taken in some areas. To the extent that supply-side regulations may require extended approval processes, policy measures that reduce vehicle-kilometres travelled can yield more immediate impacts in terms of mitigating non-exhaust emissions.

Section 4.1 describes existing measures to reduce non-exhaust emissions by lowering emission factors and by reducing vehicle-kilometres travelled. Section 4.2 discusses how policymakers can better address non-exhaust emissions from road transport and proposes a framework to guide the design of a pricing instrument addressing non-exhaust emissions specifically. Section 4.3 discusses the work that remains to be done in order to implement the proposed measures. Table 4.1 provides a broad overview of some of the measures that can address non-exhaust emissions from road transport.

Table 4.1. Policy Measures to address non-exhaust emissions

Type of measure	Scale of implementation		
	International	*National*	*Subnational/Local*

Market-based instrument		**Non-exhaust emissions charge** ^{BTRD}	**Include EVs in road pricing** ^{BTRD}
Regulation	• Standardise the measurement of non-exhaust emissions ^{BTRD} • Regulate the content of brake and tyre materials ^{BT}	• Include vehicle weight as determinant of vehicle taxes and fees ^{BT} • RBS in ICEVs ^{B} • Restrictions on studded tyre use ^{TR}	• UVARs differentiated by brake, tyre, road characteristics ^{BTRD} • Vehicle weight limitations in city centres ^{BT} • Traffic flow measures to ease congestion ^{BTRD}
Management measures			• Road washing and dust binding ^{D} • Minimise sanding and salting^{TRD}
Soft measures	• Eco-labelling ^{BTR}	• **Communication campaigns** ^{BTRD}	

Note: Measures in bold incentivise both a reduction in EFs and in vehicle-kilometres travelled. Superscripts denote non-emissions source targeted, where B, T, R and D indicate brake wear, tyre wear, road wear, and road dust resuspension, respectively. UVAR: urban vehicle access regulation.

4.1. Review of existing measures that reduce non-exhaust emissions

The "avoid, shift, improve" conceptual framework used to categorise strategies to reduce exhaust emissions from road transport (SUTP, 2019[1]) can also be applied to non-exhaust emissions.

Strategies to "improve" vehicle emission factors are covered in Section 4.1.1, which reviews the types of regulations in place and currently available technologies that reduce non-exhaust emission factors. Section 4.1.2. reviews policy strategies to reduce overall demand for vehicle-kilometres travelled through measures to "avoid" unnecessary car travel and "shift" to less emissions-intensive modes. It should be noted that many of the measures presented in these sections are not motivated by the objective of reducing non-exhaust emissions. Rather, reductions in non-exhaust emissions happen to occur as a side-benefit of other targeted objectives.

4.1.1. Reducing vehicle kilometres travelled

To date, other negative externalities associated with car use, namely those issuing from exhaust emissions and congestion, have been the primary motivation behind measures aiming to reduce the number of vehicle kilometres driven. This section presents an overview of such policies that are currently in place, although the specific features of their implementation can vary considerably. Given that these policies target all vehicle emissions and are not unique to non-exhaust emissions, there is extensive precedent for their use. This review is not intended to be exhaustive with respect to these policies; for more comprehensive reviews, see (UNESCAP, 2017[2]; CIVITAS, 2019[3]; UN ECE, 2016[4]; Rupprecht Consult (editor), 2012[5]; Rye and Ison, 2008[6]).

Reducing demand for private vehicle travel can be accomplished by a variety of measures that increase the relative attractiveness of public transport and non-motorised modes relative to private vehicles. These measures can consist of disincentives for private vehicle ownership and use, i.e. measures that raise their costs and/or inconvenience, as well as incentives for alternative modes (e.g. public transit, walking, and biking).

Disincentives for car ownership and use can be pecuniary or regulatory in nature. Pecuniary disincentives for car ownership can be one-time or reoccurring, and include purchase taxes, registration fees, and annual taxes. Disincentives for car use come in the form of operational costs, including taxes on fuel (and increasingly carbon), flat rate distance-based charges, congestion pricing, and parking pricing. Currently, few pecuniary measures differentiate the costs of passenger car use by the determinants of the marginal damages from non-exhaust emissions (e.g. vehicle weight). In some countries (e.g. Norway and Sweden), owning heavier cars is discouraged through vehicle registration fees and annual taxes that are based in part on vehicle weight (Svenningsen et al., 2019[7]). While fuel charges indirectly incentivise the use of lighter weight vehicles by virtue of the fact that such vehicles use less fuel, all else equal, explicit weight-based use charges do not exist for passenger vehicles. Although weight-based disincentives for LCV use are in place in some regions (ACEA, 2019[8]), these charges are not sensitive to other determinants of marginal damages of non-exhaust emissions such as population exposure.

In addition to increasing the cost of using private vehicles, regulatory measures can also make private motor vehicle use less convenient. Examples of regulations that seek to limit car use include urban access restrictions (e.g. low emission zones), vehicle bans based on vehicle age or technology (e.g. bans on diesel and heavy duty vehicles), license plate limitations (e.g. even/odd criteria during pollution spikes), and parking regulations, including limiting parking supply. Parking restriction policies can, for example, prohibit parking for non-resident vehicles in specific areas of the city or expand the size of the areas requiring paid parking. A time-trend analysis of parking restrictions on the number of circulating vehicles/working day in central Barcelona showed a decrease of vehicles from 2005 to 2015 of close to 10% (Querol et al., 2018[9]).

Urban vehicle access restrictions (UVARs) come in a number of forms and have been adopted by a growing number of cities in recent years (e.g. Stockholm, London, Milan, Paris and Madrid) with the primary goal of reducing congestion and exhaust emissions. In addition, the implementation of low emission zones which prohibits the access of more polluting vehicles to city centres, may have an indirect effect on the number of vehicles entering the urban area, and thus on non-exhaust emissions (Holman, Harrison and Querol, 2015[10]).While UVARs are generally motivated by the need to comply with air quality standards for NO2 and reduce congestion in city centres, an additional benefit arises from the reduction of non-exhaust emissions.

In Madrid, an analysis of the number of cars in circulation during a NOx pollution peak revealed an average decrease in traffic volume of 17% in the centre of the city when a license-plate-based access restriction was in force, and an average decrease in the NO2/SO2 ratio of 13% (Querol et al., 2018[9]). A successful case of congestion charging is that of Stockholm, where a congestion charging scheme led to a 30% reduction in circulating vehicles, as well as a 5-13% reduction in NOx and a 14-20% reduction in PM10, which is partially attributable to lower non-exhaust emissions (Johansson, Burman and Forsberg, 2009[11]; Johansson, 2016[12]).

Electric vehicles are often exempt from access restrictions and road charging schemes. In some cases, even some less emission-intensive ICEVs (such as hybrid vehicles) are exempt. The exemption of EVs and other alternative fuel vehicles from road pricing measures (e.g. in London, Oslo, Milan and Palermo) decreases the effectiveness of these schemes in reducing vehicle-kilometres, and therefore non-exhaust emissions and congestion. The effectiveness of these schemes can be particularly lower if travel demand shifts do no only occur from ICEVs to EVs, but also from public and non-motorised transport. In such cases, the net environmental benefit of the measures may be lower or even negative. Given a basic knowledge of the determinants of non-exhaust emissions, it is clear that EVs should not be exempt from restrictions or pricing instruments seeking to reduce these emissions. In Stockholm and Liverpool, for example, EVs are not exempt from city tolls. Rather than provide a uniform incentive for EV use by exempting them from measures seeking to reduce motor vehicle use, such charges should be differentiated with respect to the amount of non-exhaust emissions they produce.

Incentives to increase the uptake of public transit and non-motorised modes include improving the coverage, frequency, comfort, information provision, and payment systems of public transit services. Integrating ticketing will encourage multi-modal use of public transit systems. According public transport vehicles priority in express lanes will reduce travel times for these modes. Better quality and more well-connected infrastructure for non-motorised modes, such as protected bike lanes, sidewalks, and priority pedestrian crosswalks increase the attractiveness of using these modes in urban areas. In the long term, developing compact urban areas can also contribute to reducing the number and length of trips taken in private vehicles by reducing the distances required to access amenities (CIVITAS, 2019[3]). New forms of high-occupancy shared mobility services also have potential to reduce demand for total vehicle kilometres (ITF, 2016[13]; CIVITAS, 2019[3]).

Bike sharing programs have also been shown to be an effective strategy for increasing the mode share of cycling in urban areas, and new forms of on-demand, high occupancy public transport hold promise for drastically reducing demand for vehicle kilometres travelled (ITF, 2016[13]). The provision of park and ride facilities and incentives for carpooling can also reduce the number of vehicle kilometres driven if private vehicles are no longer used for entire trips, but instead used as a feeder to public transit. Various financial incentives, including offering subsidies for bicycle purchases, making public transport costs eligible to be tax deductible, or engaging private companies in plans to co-finance the use of public transport or soft modes for their staff, will increase the relative attractiveness of these modes.

It should also be noted that some existing measures targeting exhaust emissions could have a negative impact on non-exhaust emissions, such as freight consolidation measures to maximise the amount of freight-tonnes transported per kilometre driven. Given the importance of vehicle weight on non-exhaust emissions, is not clear whether consolidating freight will tend to increase or decrease net non-exhaust emissions. Traffic calming measures also have the potential to reduce non-exhaust emissions insofar as they reduce frequency and rates of acceleration and deceleration. Most importantly, however, the analysis and evidence gathered in this report indicate that a key strategy of urban development plans, namely indiscriminate incentives to electrify private vehicle fleets, will not be an effective strategy for reducing non-exhaust emissions.

4.1.2. Improving emission factors

Measures to improve emission factors can come in the form of regulations on materials or manufacturing processes, or in the form of pricing mechanisms that effectively incentivise the development of technologies that are otherwise cost-prohibitive. This section summarizes the range of technological solutions currently available to reduce non-exhaust emissions, the main regulations in place, as well as a number of other types of traffic management measures that can be used to lower emissions potentials. Technologies to reduce emission factors generally address the vehicle itself (brake system and tyres), but other technologies designed to reduce road wear and the resuspension of road surface particles are also reviewed in the section on management measures. Due to the high degree of geographic and temporal variability in non-exhaust emissions from road dust resuspension, measures to address them are most relevant on an as-needed basis at local scales rather than via more generalised instruments.

Regulatory measures

Vehicle lightweighting and HDV weight limitations

The evidence presented in this report indicates that vehicle weight is an important driver of non-exhaust emissions. Given that electric vehicles are considerably heavier than their conventional counterparts, vehicle lightweighting should be considered an important regulatory tool for addressing the non-exhaust particulate matter emitted by ICEVs and BEVs alike.

The mean vehicle weight of passenger cars in the EU has risen by 10% over the last 15 years (ICCT, 2018[14]) and similar trends are observed in other regions. Currently, most automobile bodies are made of steel plates, with lighter components comprising only 30% of their weight. In terms of material changes, high-strength steel and aluminum alloy have seen rapid advancements in recent years. Other lightweight materials in development include porcelain, plastic, fiberglass and carbon fiber. Lifecycle analyses can identify the potential environmental trade-offs involved in reducing vehicle weight, such as those generated by the use of more resource-intensive materials, and the GHG emissions and other pollutants released in the production and disposal of light weight vehicle parts (e.g. .

Beyond the fact that lighter weight vehicles are typically more fuel efficient than heavier weight vehicles, added incentives to produce and purchase lighter vehicles remain limited, especially for passenger cars. In several countries, however, vehicle taxes (e.g. road taxes and circulation fees) are a function of vehicle weight (ACEA, 2019[8]). Evidence indicates that while reducing the weight of passenger cars is currently cost-effective, this is not the case for LCVs (ITF, 2017[15]). Efforts to account for and internalise the benefits that lightweighting yields in terms of non-exhaust emission reductions could make it a cost-effective mitigation policy. Regulation to reduce the maximum weight allowed for vehicle type approval could also be considered (see Section 3.2).

The average load of HDVs has also been increasing in recent decades, a trend that has been driven by economic growth and the roll out of the new vehicle technologies that made such an expansion possible (Žnidarič, 2015[16]). Many municipalities have established vehicle weight restrictions for vehicles circulating in urban areas. A number of German cities implement lorry transit bans to reduce emissions and traffic. In Germany, the emissions reduction potential of circulation restrictions based on vehicle weight strongly depends on the proportion of lorries in the traffic flow, as well as on how well the measure is enforced and on compliance rates (UBA, 2015[17]). The maximum emissions reduction potentials are found to be 20% for NOx, 10% for NO2, 10% for PM10 and 7% for black soot. Across cities, dates, and compliance levels, theoretical reduction potentials ranged from 1.2-16.7%.

In Barcelona, vehicle weight limits depend on the number of road lanes. For one-lane roads, regulations prohibit vehicles weighing more than 12 tonnes during the day and more than 18 tonnes at night. On two-lane roads vehicles weighing more than 18 tonnes are prohibited during the day, and more than 44 tonnes at night. The one-lane historic city centre road is not accessible to vehicles over 7.5 tonnes. These measures have led to dramatic changes, reducing the number of heavy goods vehicles by 97.5% during cordon hours and by 97% over 24 hours. The number of heavy goods vehicles using this road per month in 2009 was less than the daily average in 2006.

Brake and tyre manufacturing regulations

The most important worldwide regulations affecting brake and tyre production are summarized in Grigoratos (2018[18]). These include the asbestos ban, the REACH and REACH-like regulations, the European regulation on the classification, labeling and packaging of chemical substances and mixtures (CLP Regulation), as well as regulations related to restrictions on the use of particular trace elements and heavy metals. Table 4.2 provides a brief overview of the regulations related to brake composition and the countries where these regulations were applied in 2018.

Table 4.2. Brake composition related regulations worldwide

Regulation	Country
Asbestos ban	Algeria, Argentina, Australia, Bahrain, Brunei, Chile, Egypt, EU*, FYROM, Gabon, Gibraltar, Honduras, Iceland, Iraq, Israel, Japan, Jordan, Korea, Kuwait, Mauritius, Monaco, Mozambique, New Caledonia, New Zealand, Norway, Oman, Qatar, Saudi Arabia, Serbia, South Africa, Switzerland, Turkey, Uruguay, USA*
REACH regulation	EU
REACH-like regulation	Albania, Australia, Brazil*, Canada, China, Chinese Taipei, FYROM, Hong Kong (China), Iceland, India, Indonesia*, Israel, Japan, Korea, Malaysia, Mexico, Norway, Philippines, Serbia, South Africa*, Switzerland, Taiwan, Thailand, Turkey*, UAE, USA*, Vietnam*
End of Life Vehicles Directive	EU
Senate Bill regulations	California (SB 346), Washington (SB 6557), Rhode Island (SB H7997), and New York State
CLP Regulation	EU

Note: Not applied in every member state or applied with some modifications.
Source: Grigoratos (2018[18]).

Trace and heavy metals content restrictions

Regulations regarding the use of trace elements and heavy metals in vehicle manufacturing have also had implications for the composition of brake systems and tyres. In Europe, the End of Life Vehicles directive requires that "Member States shall ensure that materials and components of vehicles put on the market after 1 July 2003 do not contain Lead (Pb), Hexavalent Chromium (Cr VI), Cadmium (Cd) and Mercury (Hg)" (EC, 2000[19]). The objective of the directive is to prevent waste from vehicles and promote the reuse, recycling and recovery of end-of-life vehicles and components, improving thus environmental performance for all operators involved.

Although similar federal regulations do not exist in the US, a number of US states have also adopted regulations regarding the use of heavy metals in brake friction materials. California (SB 346), Washington State (SB 6557), Rhode Island State (SB H7997) and New York State (SB A10871) have passed laws which limit the weight percentage of specific constituents within a friction material. In addition to the heavy metals that are restricted in Europe (chromium, lead, cadmium and mercury) special attention has also been paid to copper in brake systems in the US. The issue began in the early 1990s when cities south of San Francisco were having trouble meeting Clean Water Act requirements to reduce copper in urban run-off. After it was found that brake wear accounted for 35-60% of the copper in the area's urban run-off, legislators adopted measures to reduce the amount of copper used to manufacture brake systems. The legislation requires manufacturers to acquire and report third party verification of compliance level, label brakes with their compliance levels, and retest and recertify the materials used to make brakes every three years.

Table 4.3. Heavy metals restrictions enforced in California and Washington State and date of implementation for each level of regulation

	Jan 2014	Jan 2021	Jan 2025
Copper	No limit	5.0% wt	0.5% wt
Chromium (VI), Lead, Mercury	0.1% wt	0.1% wt	0.1% wt
Cadmium	0.01% wt	0.01% wt	0.01% wt
Nickel, Antimony, Zinc	Currently none – Monitored and maybe regulated in the future		

In Europe, the REACH regulation (Regulation 1907/2006) aims to improve the protection of human health and the environment through better and earlier identification of the intrinsic properties of chemical substances. This is done by the four processes of REACH, namely the Registration, Evaluation, Authorisation and Restriction of chemicals. The general principle of REACH regulation is "No data, no market". Practically speaking, this means that if a substance has not been registered for a specific application, it must not be used. Under REACH, the substances in a product must be registered if "the substances are intended to be released from the produced or imported article(s) during normal and reasonable foreseeable conditions of use." However, because the release of particles from tyres, rubber belts, brake linings and discs is not considered to be intended, no registration of their chemical components is required. An exception to this rule is products that contain Substances of Very High Concern (SVHC) in the quantity of 0.1% or more by weight, which should be reported.

One group of substances relevant to the manufacture, import, supply and use of tyres which is subject to REACH restrictions is polycyclic aromatic hydrocarbons (PAHs). PAHs include over 100 different chemicals that are formed during the incomplete burning of coal, oil and gas, garbage, or other organic compounds. PAHs are known for their carcinogenic, mutagenic and teratogenic properties and have been used in the production of tyres. REACH Annex XVII has placed a restriction on the use of 8 PAHs in tyres and extender oils (oils added to tyres during the manufacturing process).

Regulations have also been applied in countries including China, Malaysia, and Korea. Despite the similar concept of these regulations with REACH regulations, a number of differences exist regarding the timing of notifications/registrations, the threshold values applied, the availability of exemptions, and the communication in the supply chain (Grigoratos, 2018[18]).

The automotive industry has also developed tools to support compliance with the REACH regulations and the End of Life Vehicles directive. The International Material Data System (IMDS) is an on-line system used by the automobile industry that collects, maintains, analyzes and archives information on materials used for automotive components. IMDS entry is obligatory for recycling rate management, management of substances of concern, and REACH compliance. It therefore supports compliance with the obligations placed on automobile manufacturers and suppliers by national and international standards, laws and regulations, scientific findings and risk assessments, as reflected in the Global Automotive Declarable Substance List (GADSL). The GADSL is the result of a year-long global effort of representatives from the automotive, automotive parts supplier and chemical/plastics industries. The list reflects regulated substances used in the automobile and chemical industries in Japan, Europe, and the U.S. (American Chemistry Council, 2019[20]). Suppliers are required to comprehensively manage substances of concern and to make continuous efforts to reduce use in accordance with GADSL. In the IMDS system, suppliers are required to request parts and materials from automotive producers via the IMDS website, which provides a repository of up to 9,000 vehicle components and the substances they are made of.

Asbestos bans

Asbestos was historically used as a friction material in brake linings, disc brake pads, and clutch facings in vehicles because of its unique fire resistance and wear properties. Asbestos has since proven to be associated with occupational diseases, including respiratory problems (Stayner et al., 2008[21]), mesothelioma (Lemen, 2004[22]), and lung cancer (Program, 2005[23]). For this reason, the manufacturing of asbestos containing friction materials including brake pads has ceased in many places worldwide, with positive effects on ambient air quality.

In Europe, the asbestos-free directive (Council Directive 83/477/EEC on the protection of workers from the risks related to exposure to asbestos at work) was introduced with the aim of protecting workers against risks to their health arising or likely to arise from exposure to asbestos at work. In that context, many European Member States have introduced total bans in the use of asbestos for the production of brake pads, while there are still some countries where the compliance with this directive has not been verified (IBAS, 2019[24]).

In the United States, brake friction materials are not covered by US Environmental Protection Agency asbestos regulations at the federal level. Legislation has been enacted in California (SB 346) and Washington (SB 6557) to eliminate the use of several toxic chemicals, including asbestos, in brake pads, drum linings, and heavy-duty brake blocks since 2014. More recent bills have prohibited the production, import, transfer, provision, or use of asbestos or any material containing more than 0.1% asbestos by weight. In practice this constitutes a total asbestos ban because 0.1% is below the detection limit of existing measurement methodologies. Despite the global trend towards total asbestos bans, some countries including China, Russia, India and the United States still permit the use of asbestos in some applications (IBAS, 2019[24]).

Studded tyre restrictions

Many countries regulate the use of studded tyres, although this is generally to reduce expenditure on road maintenance rather than the damages associated with non-exhaust emissions. Restrictions come in a number of forms, including restrictions on the type of tyres that can be used (e.g. Spain, where tyres with studs above 2mm are prohibited), the speeds that can be driven using studded tyres (e.g. Ireland, where the maximum speed allowed for cars with studded tyres is 96-112 km/hr depending on the roadway), the conditions under which they can be used (e.g. Turkey, where they are only allowed for use on ice-covered roads), and the vehicles which can use them (e.g. in Austria studded tyres are only permitted on vehicles <3.5t GVW).

Many countries combine various criteria and apply them on a seasonal basis. Switzerland allows the use of studded tyres on vehicles <7.5t GVW in winter-spring months (max. speed 80 km/h) on snow-covered roads. Norway, Denmark, Lithuania and Sweden (with some exceptions) allow them during winter-spring, but Norway levies a charge for their use. France restricts the use of studded tyres to vehicles <3.5t GVW and to winter-spring months allowing a maximum speed of 90 km/h.

As a regulation targeting a specific technology, these measures appear to be effective. In Hornsgatan (Sweden), a combination of reduced traffic volume and reduced use of studded tyres reduced road wear by an estimated 72% (Norman et al., 2016[25]). Given that studded tyres constitute an important safety measure in the countries where they are used, and given that their use is generally less prevalent in urban areas where population exposure is high, their relevance as a measure to target non-exhaust emissions in these areas is limited.

Available technologies

Regenerative braking systems

Motors in electric cars are designed not only for acceleration, but also for deceleration. When the brake pedal is pressed in the generator mode, the motor works in the reverse direction in order to recharge the battery. Regenerative braking systems (RBS) increase the range of BEVs as well as the fuel economy of HEVs by recuperating some of the energy normally lost as heat when braking using conventional friction brakes. An additional benefit of regenerative braking systems is that they significantly reduce brake wear emissions. While regenerative braking can be sufficient for most daily use brake applications, the combined use of regenerative braking and friction braking will remain necessary in the near future for safety reasons (AUDI AG, 2016[26]), as emergency stops still require the use of the latter.

Estimates of the brake wear emissions reductions made possible by RBS range from 25% to 100%. Regenerative braking has been reported to reduce brake wear by 25-50% (Platform for Electromobility, 2016[27]). Van Zeebroeck and De Ceuster (2013[28]) assume that regenerative braking reduces the PM emissions associated with brake wear by 50%. Timmers and Achten (2016[29]) assume zero brake wear emissions for electric vehicles, i.e. a 100% reduction. Hooftman et al. (2016[30]) state that EVs require about two-thirds (66%) less friction braking action than ICEVs due to regenerative braking. Their analysis is based on the service times of brake pads on Teslas, BMW i3s, and Nissan Leafs, which on average last roughly two-thirds longer than those on diesel/petrol vehicles. They note that the savings made possible by regenerative braking outweighs the additional wear due to the higher mass of electric vehicles. Barlow (2014[31]) suggested that regenerative braking produces virtually no brake wear, and (Ligterink, Stelwagen, & Kuenen (2014[32]) assume regenerative braking reduces wear by up to 95%. Based on a report by Althaus and Gauch (2010[33]), Del Duce et al. (2014) report that brake wear emissions are 80% lower for EVs than ICEVs. Nopmongcol et al. (2017[34]) assume a 25% reduction in brake wear with the use of regenerative braking. Observed reductions in particle number concentrations range from 66-99% (Augsburg and Hesse, 2018[35]).

Although regenerative braking systems are one of the most promising technologies for reducing non-exhaust emissions from brake wear in terms of their cost-effectiveness, they are currently only used in electric and hybrid-electric vehicles. Therefore, although electric vehicle manufacturers are constantly developing more technologically advanced and light weight RBS, the penetration of these systems in vehicle fleets will grow only as fast as that of electric and hybrid-electric vehicles. For this reason, industry should be encouraged to explore the extent to which RBS could be easily incorporated in ICEVs, e.g. via kinetic energy recovery systems. However, some trade-offs of RBS have been pointed out. If friction brake systems on vehicles with RBS are slow to reach bedded conditions, this could result in increased brake wear emissions when they are used. Additionally, if brake disc pads are degraded with the little use they receive in conjunction with RBS, rusted surfaces could lead to poor bedding conditions and higher brake wear.

Friction braking systems

While regenerative braking and driver assistance systems are aimed at preventing brake wear emissions by minimising the use of friction braking, a number of measures aim to reduce the amount of particle wear produced from the friction and heat that occur during the braking process itself. These technological measures mainly consist of physico-chemical modifications of the composition of rotors and pads, although the design of the brake system (geometry, ventilation, clamp, etc.) is also important. Composition changes should be evaluated carefully since they may involve trade-offs in terms of the types and relative toxicity of particles generated: for example, one modification may reduce the mass of particles emitted but increase their number, or may reduce the metallic particulates but increase the organic content of PM.

The EU funded LOWBRASYS project aims to test a novel and low-emission brake system that will reduce micro and nanoparticles from brake wear by at least 50% (Kousoulidou, Perricone and Grigoratos, 2019[36]). The project addresses both particulate emissions in the typical micrometer ranges, which are important for PM *mass* reduction, as well as ultrafine particles of less than 0.1 micrometers, which are important for particulate *number* reduction. A number of different technologies are considered, the most well-developed of which are the use of novel material formulations for brake pads and discs, the design of environmentally friendly braking strategies that optimise vehicle braking action, and technologies for capturing PM particles near their source. The interaction of new surface combinations must be rigorously tested and a clear, transparent procedure should be followed to ensure that the use of new braking materials do not have negative impacts on human health. For this reason, the LOWBRASYS project received a 6 month extension in order to explore further configurations of new pads and execute additional toxicity tests (Kousoulidou, Perricone and Grigoratos, 2019[36]). The LOWBRASYS technology is a candidate to become Best Available Technology (BAT) and serve as a basis to support the preparation of robust measurement standards and legislations on non-exhaust PM emissions, which is not addressed by any current EU legislation.

Box 4.1. The LOWBRASYS project

The LOWBRASYS project tests a novel and low-emission brake system designed to reduce micro and nanoparticles generation by at least 50%. The main objectives of the project are to:

1. Develop novel environmentally friendly material formulations for brake pads and discs used in passenger cars.
2. Develop software functions for dynamic assisted braking strategies and develop a human-machine interface (HMI) smart dashboard.
3. Develop new technology for capturing particles near the braking system.
4. Use life cycle costing, life cycle assessment and comprehensive cost-benefit analysis to assess the financial costs and environmental benefits of proposed new material formulations vs. existing formulations.
5. Pursue an improved understanding of brake wear emissions on the basis of particle numbers vs particle mass concentrations.
6. Environment: the project had demonstrated to have the potential to improve air quality (lowering PM emissions), prevent ecosystem impacts and waste, and increase resource efficiency.
7. Propose a roadmap for the development and implementation of brake wear emissions regulation in Europe, emphasising the need to prioritise the development of a methodology for sampling and measuring brake wear particles followed by a testing period in order to accurately measure real-world brake wear emissions.

The project considered a series of different technologies, the most important being the introduction of novel material formulations for brake pads and discs, the development of environmental friendly braking strategies that optimise the vehicle braking action and reduce the particle emissions and the development of breakthrough technology for capturing particles near the PM source. The project concluded that no single technology was able to reduce PM emissions by 50%. Rather, a multitude of different technologies was needed to achieve this reduction. A conclusion of the project is that further development is required to advance the proposed solutions from prototype-level technologies to market ready applications. In this context, the UNECE- GRPE PMP IWG recognizes the need for prioritizing the development of a suitable methodology for sampling and measuring brake wear particles followed by a testing period in order to accurately measure real-world brake wear PM and PN EFs.

Source: Kousoulidou, Perricone, & Grigoratos (2019[36])

Brake disc composition, coating and treatment

Brake discs made of gray cast iron represent the majority of brake rotors produced today because their manufacturing process is well understood, but most of all, they are very inexpensive to produce. Some improvements can nevertheless be made to reduce the brake wear and particle emissions they generate.

Some manufacturers have discussed adding titanium in the range of 0.1%-0.25% by weight to high-carbon cast iron brake discs to improve their hardness and reduce wear (Sichuan Province Vanadium and Titanium Brake Ltd., 2014[37]). Molybdenum has also been suggested as an additive to cast iron rotors to increase their wear resistance. Early evidence suggests that an addition of up to 1.5 wt% to the iron cast is feasible and results in wear improvement without affecting other characteristics (Piwowarsky, 2013[38]). Patterson, Standke, & Kocheisen (2013[39]) have also found that additional molybdenum reduces wear. Shangdong Hong MA Construction Machinery Co. (2014[40]) has presented a new disc material composed of a high-carbon, low-silicon cast iron containing niobium. According to the inventors, increasing niobium by 0.1-0.3 wt% improves strength at high temperatures and resistance to wear. However, these materials have not been widely introduced to the market because the addition of titanium or molybdenum increases production costs, which, under current policy conditions, is difficult to justify in the manufacturing of braking systems.

Gray cast iron disc coatings

Disc coatings are a very promising mitigation measure from a technological point of view, but the additional costs entailed (EUR 100 or more per disc) currently prohibit their market entry. Mayer and Lembach (2012[41]) describe a layer concept to improve the wear resistance of brake discs. A protective layer is made of an iron-based matrix alloy (chromium or nickel), as well as hard materials (carbides or ceramic oxides). Between the cast iron base body and the protection layer, a so-called sealing layer helps bond the friction surface layer. Mayer and Lembach (2013[42]) also discuss the use of a 10-55mm thick anti-wear layer on the rubbing ring of a brake rotor composed of a metal alloy or ceramic material. An integrated wear indicator could alert drivers when this protection layer is worn out. The indicator material would be made of a colored, black or white ceramic material, or a woven or braided fabric and could also consist of mineral pigments. A wear-indicating element in combination with a wear-reducing surface coating is also presented in Dupuis (2013[43]). In this case the wear indicator consists of a raised portion between the friction ring and the surface coating that runs around the ring in the form of a wavy line. These indicator proposals could be very useful in optimising the wear of brake rotors, which are characterised by low mass loss due to hardened surface coatings, making it difficult to gauge the extent of their wear.

Qihong (2011[44]) present a cast iron disc with a surface-hardened layer composed of hard ceramic particles and a nodular cast iron. Although its thickness of 2e5 mm could lead to crack or delamination processes over lifetime, the patent for the disc contends that it enhances the rotor by a factor 4.6. Another coated brake disc patent is presented by Keith (2009[45]), who introduces a dual-coating to enhance the wear performance of the rubbing ring as well as the corrosion prevention of the hat of the rotor. The rubbing ring is first prepared by roughening of the surface followed by the deposit of a 20-30 mm thick bond coat layer of adhesion-promoting material composed of nickel or a nickel-alloy. Next, a wear-reducing friction surface coating is applied. The layer can be made of pure alumina material or alumina alloy material. The finished layer thickness should be in the range of 100-400 mm, preferably between 150 and 250 mm. Finally, the rotor hat is also coated to protect it from corrosion. Özer and Lampke (2012[46]) also describe a thermal spray coating material to wear reduction purposes. A ceramic powder (30-35 wt%) and a metallic powder mixture (65-70 wt%) are considered. The ceramic powder is composed of reinforced particles of aluminum oxide, zirconium oxide, and titanium oxide, while the metallic powder mixture consists of zinc alloy (25-30 wt%) and iron alloy (70-75 wt%). These proposed spray coatings are designed to be economical and extend the service life of the cast iron base body. Lembach and Mayer (2012[41]) also describe design of a wear-reducing surface coating, the thickness of which varies according to the cooling channel and fin design. Because the base body material and the coating are characterised by different thermal expansion coefficients, mechanical strains are likely to occur when a coating of uniform thickness is used. This can lead to cracks in the coating or even to delamination, and cause hot spots that lead to brake judder. The authors therefore recommend a thicker coating in the area of the cooling channels where the thermal expansion is smaller.

For commercial vehicles, Khambekar et al. (2004[47]) describe a non-ceramic metallic coating for application to cast iron brake rotors and its production method and provide a number of recommendations regarding the cast iron composition of the base body. The metallic character of the layer optimises adherence to the cast iron base body relative to ceramic composites. The authors recommend a roughening of the friction ring surface as a pretreatment, after which the coating is applied via a thermal spray coating method that leads to reactions of the metallic elements with the cast iron. This reduces not only general wear, but also wear that occurs at high temperatures. A preferred composition of the coating consists of 16% of Chrome and a steel alloy. In addition, the composition of the base body of the rotor should be modified in the way that a molybdenum content of 6-8% and a chrome content of less than 5% are considered. The authors predict that service life will double with the use of a 0.7mm thick coating. For a brake rotor with a diameter of 430 mm and a weight of 25 kg, this would require 250-450 grammes of coating composite per rubbing ring.

Treatments for gray cast iron rotor discs

In addition to coating methods, ferritic nitrocarburizing (FNC) is a thermochemical treatment method that is also designed to protect cast iron rotors from corrosion and to increase their hardness in order to improve wear performance. The application of the FNC method is detailed in Hanna (2009), among others, and Riefe et al. (2011[48]) have found that it reduces brake wear. FNC-treated brake rotors currently have a very high market penetration among passenger cars in North America, but other markets remain dominated by the use of conventional gray cast iron rotors. The FNC method involves no coating, rather it enriches the disc with nitrogen and carbon and uses a “metallic” diffusion layer and “ceramic” compound layer. The diffusion layer is dominated by iron-nitrides and is four to five times harder than the gray cast iron basic material and the compound layer is characterized by the generation of nitrides and carbides and a hardness factor that is five times higher than cast iron.

FNC works well with adhesive friction mechanisms, which means that non-asbestos organic (NAO) brake pads are the best for use in combination with FNC rotors (Riefe, Holly and Learman, 2011[48]). However, it should be noted that coefficients of friction have been found to fall by 10%-15% using FNC (Riefe, Holly and Learman, 2011[48]; AUDI AG, 2016[26]). The use of low steel or UNECE-compliant brake linings would mean higher friction levels, but greater wear, removing almost 80% of the compound layer after 2000 km. FNC treatments cost less than coatings (approximately EUR 10 per disc). Although coatings and treatments such as FNC appear to be the most promising approaches for improving brake wear, the additional manufacturing costs they entail remain a barrier for further market penetration.

Aluminum Brake Discs

Although gray cast iron brake rotors have many advantages, other base materials have gained attention for automotive applications in the last 20 years. Brake discs made of aluminum represent a promising technology for future brake systems because they reduce not only weight but also mechanical wear. They are, however, at least EUR 50 more expensive than gray cast iron discs.

Özer (2009[49]) describes an aluminum metal matrix compound (Al-MMC) brake rotor composed of a metal matrix made of AlSi2O with 15-25 wt% silicon carbide. A spray-forming adds other hard particles 3-50 mm in diameter, which can be made of Al2O3, SiO2, TiO2, SiC, tungsten carbide, boron carbide, titanium nitride, or/and titanium diboride. In total, the percentage of hard material particles is 45 wt%. Lampke and Özer (2013[50]) provide more details on the production process of this type of aluminum brake rotor. Hino and Miyashita (1984) also propose an aluminum brake rotor designed to have high abrasion resistance and generate less brake pad wear, which is accomplished through the addition of a ceramic antiwear material and/or a solid lubricant to an aluminum hypereutectic silicon alloy. The ceramic material consists of Al2O3, SiC, or SiN4 and comprises 3-20 wt% of the rotor. A solid lubricant such as graphite could also be added, though in an amount not exceeding 5 wt%.

Lower surface roughness has also been identified as strategy for reducing brake pad wear. To this end, Lampke and Özer (2012[51]) propose a brake rotor base body made of aluminum in combination with a prefabricated steel wear protection layer matching the dimensions of the friction ring. This combination allows for improved wear resistance, corrosion resistance, and thermal load carrying capacity. Smolen (1991[52]) also proposes an aluminum rotor with a ceramic protection layer that is applied to the braking surface at a thickness of 0.33-0.50 mm. The coating consists of aluminum oxide, aluminum titanium, and magnesium zirconate. Because the coating acts as a heat reflector, it prevents the rotor from attaining excessive temperatures that could crack or disintegrate the disc. In addition to improvements related to thermal performance, the ceramic coating is also expected to decrease wear.

Rotors made of aluminum, especially reinforced Al-MMC, constitute a very promising approach to reduce brake wear and particle emissions. Even if issues surrounding their material and manufacturing costs remain (e.g. the critical temperature limit of around 400ºC), Al-MMC rotor concepts are expected to become increasingly relevant in the future due to their lighter weight and lower brake wear. The ongoing market penetration of hybrid and full electric vehicles could help overcome the drawbacks associated with their low temperature limits, because regenerative braking generally lowers the maximum operating temperature reached when friction braking is employed.

Ceramic Brake Discs

Ceramic composite brake discs, also known as ceramic brake discs, are another technology that can reduce wear and brake particle emissions. Originally designed for high-performance applications, this type of brake rotor has also demonstrated significantly better wear performance due to its high hardness (Kienzle and Kratschmer, 2007[53]; Johnson, 2009[54]; James, Murdie and Walker, 2002[55]). In terms of their design, they are comprised of a multilayer structure consisting of carrier bodies and friction layers that are both reinforced with carbon fibers. Although a great deal of effort has been expended to improve the manufacturing process of ceramic brake discs, their production is still very time- and energy-consuming, and the level of automation is lower compared with that of cast iron discs, which leads to significantly higher production costs (400 € to 600 € per disc). For this reason, although ceramic brake rotors can be an effective technology for reducing brake wear emissions, their future use is expected to be limited to niche applications such as high-performance sports cars.

Titanium Brake Discs

Other research has shown that titanium also has advantageous properties as a brake rotor material. Muthukannan, et al. (2014[56]) discuss the use of titanium made brake rotors (laser nitrided Ti-6Al-4V, Ti-LSN) and compare the performance of commercial gray cast iron with untreated titanium alloy. They find that untreated titanium alloy exhibits significantly higher wear rates and volume losses, presumably due to poor resistance to abrasion processes and severe plastic deformations. Laser treated titanium material, in contrast, shows a much better wear performance in all operating conditions and its frictional behaviour is comparable to that of cast iron. No adapted lining material was used in this test, suggesting that wear-reducing potential may be even greater than that measured in the study. However, similarly to ceramic brake discs, the use of titanium material remains limited, mostly appropriate for niche applications because of the high material and manufacturing costs. Given that the technology is currently in the pre-development stage, no estimation of their manufacturing costs is currently possible.

The EU-funded REBRAKE project indicates that it is possible to design a disc brake system for a European standard car that reduces PM10 emissions by at least 32% using a standard European pad and a heat-treated rotor. A further reduction to 65 wt% PM10 emissions can be achieved with NAO pad material and a heat-treated disc. The treatment consists in heating the disc to 860 °C for 2 h under controlled atmospheric pressure. Subsequent oil quenching and tempering for 4 h at 180 °C increases the hardness of the cast iron from an initial 210 ±10 Brinell hardness number (HB) to 473 ±25 HB.

Brake pad composition

The second friction partner, the brake pad, also represents a source for remediation measures to reduce brake wear emissions. Because the pad is always considered as the weak partner of the friction couple, most approaches focus on increasing the hardness and mechanical strength of the lining material. Currently, additional costs for these approaches are not expected to exceed 10 € to 20 € per pad.

Given the recent implementation restrictions on the use of heavy metals in brake pads in California (Senate Bill (SB) 346), Washington (SB 6557), Rhode Island State (SB H7997), and New York State (SB A10871), as well as further restrictions foreseen in the coming years, OEMs and after-market manufacturers are in the process of eliminating the use of a number of toxic chemicals (Cu, Cr (VI), Pb, Hg and Cd) in brake pads, drum linings, and heavy-duty brake blocks. Nickel, antimony and zinc could be affected by similar restrictions in the future. Several studies, such as the REBRAKE project, have however found that eliminating or reducing the copper content of brake pads reduces their friction coefficient and increases particulate emissions. A potential substitute for copper is Barite, which is normally used as a filler and supposedly non-hazardous (Konduru et al., 2014[57]; Loza et al., 2016[58]). Barite ore is commonly available, easier to extract, more cheaply mined, and more quickly processed than many other commodities.

Zhao (2013[59]) proposes a brake pad composed of composite ceramic fibers that are designed to be soluble, decomposable, and can be used as reinforcing fibers. The authors emphasize that the fibers do not affect human health and generate low environmental impacts. They also state that the use of a soft ceramic fiber protects the rotor disc from excessive wear. Similar to other brake pad designs, the composite ceramic brake pad consists of a lining matrix and a backing plate made of steel. Bowei (2009[60]) discusses a non-metallic ceramic brake pad for automotive applications, recommending the use of a ceramic binder (10-20 wt%) with a modified aluminium sodium silicate to improve heat resistance and wear in high temperature operating conditions.

Limited attention has been paid to the release of organic compounds from brake pads, despite the fact that these compounds represent a significant part of brake pad formulations (Plachá et al., 2017[61]). The high temperatures and pressures on involved in friction braking cause a thermal decomposition of the organic and carbonaceous components of brake pads, or thermal fade (Yun, Filip and Lu, 2010[62]). The thermal degradation of organic (polymer matrix) brake pads begins at approximately 150ºC, and maximum mass loss due to the oxidative thermal degradation of organic components occurs above 300ºC (Křístková et al., 2004[63]; Kukutschová et al., 2010[64]), when thermally less stable components of brake pads (e.g., phenolic resin, rubber, graphite, coke) interact with ambient gases and oxygen (Kukutschová et al., 2009[65]) or undergo pyrolysis (Plachá et al., 2017[61]). This process releases very fine amorphous carbon particles and volatile organic compounds, including formaldehyde, that have significant implications for human health (World Health Organization, 2000[66]; WHO, 2006[67]). Metals and their oxides can also catalyze the degradation process of phenol-formaldehyde resin (Křístková et al., 2004[63]). Nano-vermiculite has been proposed as a replacement for phenolic resin as binder in order to enhance the thermal stability and wear resistance of brake pads. Another advantage of vermiculite is that it is currently considered to be a non-carcinogenic material, as it has not yet been shown to have an effect on human health.

Recent findings also shed light on the potential of a new generation of reinforced mineral fibers with varying lengths and aspect ratios (length/diameter) to reduce brake disc and pad wear (Santamaria Razo and Persoon, 2016[68]). An increase in fiber length is associated with a decrease in friction, but an increase in frictional stability. This leads to decreased disc wear and decreased vibration when braking, and the longer the mineral fibers, the better the wear performance. These new fibers have been shown to operate properly under nearly the entire range of temperature conditions which imply improved wear performance.

In summary, changes to brake pad composition have large potential for brake wear reduction. Given the wide range of materials available and the possible trade-offs entailed by different compositions (e.g. with respect to friction performance, noise, vibration, etc.), brake pad options deserve careful consideration on a case-by-case basis. Because every vehicle build has unique design and performance requirements, a general and market-wide adoption of any one specific brake pad composition is unlikely (Gramstat, 2018[69]).

Brake dust capturing systems

Another way to reduce brake wear emissions is to install specialised devices on vehicles to collect emitted particles. Brake particle collection systems have not been introduced to the market, neither from OEMs (original equipment manufacturer), nor as aftermarket solutions. Brake particle collecting systems patented so far include absorbers, adhesives, magnets or collector devices that use the rotational air flow of the rotor or additional fans. Several experimental studies have already been carried out with promising results.

The LOWBRASYS Project is working on a technology for capturing particles near the brake system in order to reduce PM10 emissions by 10% (Kousoulidou, Perricone and Grigoratos, 2019[36]). However, the efficiency of this technology with regard to ultrafine nanoparticles is unknown and practical uncertainties remain regarding its production on a large scale. The limited amount of space generally available in the wheel and wheel house also limits the possibilities for introducing additional components such as fans, impellers, pipes, and filter boxes. Interactions of the collecting devices with the brake are also possible, especially at the high temperatures reached during braking events (up to 700ºC), which could impact the collecting devices. Last but not least, possible side-effects regarding noise and vibration should be considered. Rotating parts, especially with direct (frictional) contact with the brake rotor, might cause or at least aggravate noise and vibration issues, such as brake squeal or brake judder. Although standardised costs are not yet available for brake particle capturing systems, they are likely to fall in a range of EUR 50-100.

Box 4.2. Lead wheel weights

Lead wheel weights (LWW) are attached to the rims of automobile wheels in order to balance the tyres. These weights can come loose and fall off. LWW that fall from vehicles can be abraded and ground into tiny pieces by vehicle traffic resulting in higher lead content in fugitive dust along urban roadways, a potentially significant source of human lead exposure in urban environments (Root, 2000). Most wheel weights enter use via commercial tyre dealers and automotive repair and maintenance shops (USGS, 2006[70]; ECCC, 2013[71]).

Several countries have made it illegal to sell, manufacture or install LWW and some major tyre manufacturers and leading retail organizations have voluntarily removed LWW from their operations or are transitioning away from lead. Automobiles imported into Canada from Japan and the EU are equipped with non-lead wheel weights, consistent with the manufacturers national requirements (Government of Canada, 2017[72]).

Despite regulatory action in Europe and by some US States, and a decade of voluntary initiatives encouraging the use of alternatives, LWWs are still used in significant quantities in several countries as for example in North America. There have been various voluntary initiatives to reduce or eliminate the use of LWWs in North America. For example, the US EPA launched its 'National Lead-Free Wheel Weight Initiative' in August 2008 (US EPA, 2008[73]). It was developed as "a partnership among federal agencies, states, wheel weight manufacturers, retailers, tyre manufacturers, automobile trade associations and environmentalists" to encourage "the transition from the use of lead for wheel weights to lead-free alternatives." The Tire Industry has also developed an Environmental Best Practices report which describes procedures to follow in transitioning away from LWWs and the precautions that should be followed to ensure the proper handling, management and recycling of existing LWWs (TIA, 2008[74]).

The most common alternative material is steel, but other non-toxic alternatives are also being used, including high-density polymers in specialty applications, and aluminum and zinc alloys. Costs for non-lead wheel weights would comprise a small percentage of the costs of new vehicles and new tyre purchases. According to the California EPA, lead or zinc wheel weights lost on the roadway have much higher potential impacts to human health or the environment compared to steel (California EPA, 2011[75]). The substitution of zinc for lead weights poses a burden shift as the losses during use are more harmful to the environment than lead. Considering the assumptions outlined above, the impacts from lead- or zinc-based wheel weight losses to roadways, greatly exceed their manufacturing impacts. Therefore, steel appears to be the preferred alternative for clip-on weights due to its comparatively low toxicity and reasonable environmental impacts from manufacturing activity.

A transition to steel weights would raise the total costs of wheel weights deployment by about 0.25%. On the basis of an economic cost-benefit analysis of a potential lead weight phase-out, Campbell et al. (2018[76]) concluded that the social benefits from a regulated requirement for non-lead wheel weights exceeded anticipated costs to the industry.

Source: USGS (2006[70]), ECCC (2013[71]), Government of Canada (2017[72]), US EPA (2008[73]), TIA (2008[74]), California EPA (2011[75]), Campbell et al. (2018[76]).

Driver Assistance Systems

Evidence suggests that smooth traffic flows with fewer stop-and-go patterns generate fewer non-exhaust emissions than more congested traffic flows (Gruden, 2008[77]). Driver Assistant Systems (DAS) can give drivers feedback with the aim of increasing the efficiency of their driving style, making these systems relevant for the reduction of brake and tyre wear emissions. The more these systems can intervene to increase efficiency without requiring action on the part of the driver, the greater their assumed positive impacts on efficiency (Guckeisen, 2018[78]). A number of technologies are currently in use by original equipment manufacturers (Gramstat, 2018[69]; Guckeisen, 2018[78]) but no studies exist quantifying their effectiveness in terms of reduced non-exhaust emissions. For example, the predictive efficiency assistant (PEA) uses GPS-system data (topography, bends, speed limits, etc.) and vehicle sensors (cameras, radar, etc.) to inform the driver in advance about an upcoming need to decelerate.

The LOWBRASYS project implemented software functions for dynamic assisted braking on passenger car vehicles to develop a human-machine (HMI) smart dashboard interface. Adaptive cruise control (ACC) autonomously maintains a constant speed using longitudinal dynamic control. Under ACC, deceleration is preferentially carried out by free-wheeling, motor braking, or recuperation (in the case of an RBS) rather than friction braking, and acceleration is used on an as-needed basis. Lane departure warning (LDW) is a mechanism designed to warn the driver when the vehicle begins to move out of its lane on freeways and arterial roads (unless a turn signal is on in that direction). These systems are designed to minimize accidents by addressing the main causes of collisions including driver error, distractions and drowsiness. Adaptive navigation systems record data during driver's typical daily commute to learn routes, road gradients, curves, and breaking points. The system also uses a camera to visualise the roadway. Once the system learns the route, it can optimize power use to be more efficient during the trip, helping to reduce emissions and improve fuel economy.

All these types of DAS significantly reduce the use of friction braking. Rychlak (2010[79]) examines how DAS can optimise driving through a curve. This requires a measurement of the curve characteristics and the calculation of a drive scenario that reduces CO2 emissions, tyre abrasion and brake dust. A number of technologies are required for this task. In addition to "environment-to-car" communication, "car-to-car" communication is also required to complete the data transfer factor and optimize driving (Integreen, 2012[80]; UR:BAN, 2013[81]). It is expected that the combination of sophisticated assistance systems (ACC, active curve assists and lane departure assists), together with a sophisticated data transfer (Car2X communication) comes close to the requirements needed to enable autonomous driving and can lower brake wear emissions insofar as it limits the usage of friction brakes only for emergency braking or long mountain descents. Importantly, drivers must also feel comfortable using DAS to improve safety and reduce emissions manufacturers (Gramstat, 2018[69]; Guckeisen, 2018[78]). Estimating the costs of DAS is a complex task due to the variety of sensor systems available (e.g. on board vs. off board), but driving feedback applications could be implemented quite easily and cost-effectively (EUR 10-20 per application) (Gramstat, 2018[69]).

Local-level traffic and road management measures

Traffic flow measures

Traffic speed limitation is a common policy measure to improve safety as well as to reduce emissions from vehicles both on highways and in city centres. While its effects on air quality have been studied mostly in the context of exhaust emissions, there is some evidence on its effectiveness in reducing non-exhaust emissions. In Berlin, lowering the speed limit from 50 to 30 km/h, combined with effective enforcement, resulted in a decrease in traffic-related PM concentration of 25%-30% (and 5% of total ambient PM concentration). The traffic-related NO2 concentration at the curbside also fell by 15%-25% and the total concentration fell by 7%-12% (Lutz, 2013[82]). These results were achieved by synchronizing the traffic lights to ensure smooth traffic flow at 30 km/h (i.e., at closer to a constant speed rather than an average of stop-and-go speeds) (Casanova and Fonseca, 2012[83]). According to the Copert emission model, in contrast, which uses average vehicle speeds, reducing average traffic speed from 50 to 30 km/h increases both exhaust PM and NO2 emissions (Int Panis et al., 2011[84]).

A drawback of speed limitation policies is that they require a high level of enforcement (European Commission, 2019[85]). A DSA called Intelligent Speed Assistance (ISA) could be an important in-vehicle measure for tackling speeding. The EU-funded project SpeedAlert has been working on the definitions, classifications and standardisation requirements for a European application of a speed limit information and warning system.[2] The SafeMap project (in the framework of the French-German DEUFRAKO-programme) is also working on a feasibility study and a field trial with a system to inform drivers of a safe speed at a particular location (T'Siobbel, 2005).

Insofar as it reduces travel times, however, improving traffic flow may induce additional demand for travel by private vehicles in the long run. This risk should be managed in an integrated manner in mobility plans to ensure that the expected air quality benefits of traffic management measures continue to be delivered in the long term.

Measures to reduce road dust resuspension

Chemical dust suppressants have been used on unpaved roads and in the minerals industry to suppress dust for a long time in some countries. Since the 90s, they have also been used on paved roads in Norway, Finland, Sweden, and Austria (among other countries), both in tunnels and on open roads. Dust suppressants are sprayed onto the road surface, where the particles that come into contact with it are bound, reducing their capability of becoming airborne when agitated by the wind, tyre action, or vehicle turbulence. Dust suppressants also lower the freezing point of precipitation.

Many different types of dust suppressants have been tested around the world. In Europe, the focus has been on the use of hygroscopic compounds that absorb water when relative humidity exceeds 50%, namely magnesium chloride, calcium chloride, calcium magnesium acetate (CMA), and potassium formate. CMA has been the subject of tests on paved roads in Sweden (Norman and Johansson, 2006[86]), Austria (AIRUSE, 2016[87]), Germany (Reuter, 2010[88]) and the United Kingdom (Barratt et al., 2012[89]). Other studies have been conducted in Sweden, Finland, and Norway (Gustafsson, 2013[90]; Gustafsson, Jonsson and Ferm, 2010[91]). In general, effective dust suppression has been shown to require repeated applications and treatment over large areas. In Sweden and Austria, where road dust emissions represent a major contributor to PM10 levels, the application of CMA led to a decrease of the daily mean PM10 by up to 35% on a highway, and 5-20% in the city, with decreases of up to 40-70% measured during certain hours. The studies carried out in Germany and the United Kingdom could not detect a significant PM10 decrease on typical urban roads. When decreases in PM10 decrease are found, they tend to be short lived (lasting a few hours), which suggests that the effectiveness of CMA in binding deposited particles is closely related to the degree of road moisture.

In Barcelona, the application of CMA and magnesium chloride on a typical urban road had no significant impact on air quality (Amato et al., 2014[92]). In contrast with Sweden, Austria and Norway, the ineffectiveness of these measures in Mediterranean regions is likely due to faster decreases in moisture from the evaporation induced by higher levels of solar radiation. Böhner et al. (2011[93]) have also indicated that road dust binding agents are only effective when the relative air humidity is at least 35%. In addition, it is noteworthy that the sites where CMA or magnesium chloride were found to have an effect were characterized by very high road dust loads (studded tyres, road sanding, and industrial dust).

Box 4.3. On-board dust filtering

A recent invention to address road dust emissions comes in the form of an active filtering system installed in the undercarriage of a vehicle. This system has been deployed in electric vehicles by the Deutsche Post DHL. The filter is installed in the lower part of the body at the level of the rear axle, so that no valuable cargo space is lost, and fans are installed behind the filters, enabling the vehicle to filter fine dust from the ambient air even when idling. Particle filters are equipped with sensors to monitor their efficiency. Information is collected on filtration performance, the volume of purified air, the concentration of particulate matter and meteorological data. The data can then be sent to the cloud with a web interface for evaluation.

Source: Heise Online (2017[94])

Street sweeping

Evidence indicates that street sweeping alone (without water) does not improve air quality in the short term due to the inability of sweepers to effectively catch particles of diameters below 10 µm (Amato et al., 2009[95]). However, this does not mean that in the long term street sweeping is not beneficial for air quality, due to the removal of larger particles, which could eventually be ground into smaller particulate matter. Vacuum sweepers are preferable to simple mechanical sweepers. However, because road sweepers have the potential to increase local PM10 concentrations due to the resuspension induced by the operating vehicle itself, vacuum sweepers should be equipped with water spraying and particle filters at the vacuum air exhaust.

Street washing

Water on road surfaces reduces the mobility of road dust, making street washing a potentially effective measure for mitigating non-exhaust dust resuspension. When water adheres to deposited particles, it increases their mass and surface tension forces, decreasing their likelihood of becoming airborne, especially as the cohesion of wetted particles often persists after the water has evaporated due to the formation of aggregates.

The effectiveness of street washing in reducing road dust has been quantified in a number of studies worldwide. In Düsseldorf, street washing twice a week reduced daily mean PM10 concentrations by about 2 mg/m3 on a busy road (John et al., 2006[96]). In Stockholm, street washing of roadside next to a highway ramp in favorable weather reduced road dust concentrations by 6% (Norman and Johansson, 2006[97]). Additionally, while the control site exceeded 50 mg/m3 on 12 occasions, the test site exceeded this amount on 10 occasions. Street washing was also found to be beneficial in the Netherlands, where its effectiveness varied depending on local conditions such as pavement material, weather conditions, and particle solubility (Keuken, Denier van der Gon and van der Valk, 2010[98]). In Madrid, Karanasiou et al. (2011[99]) found that the mass contribution from road dust was 2 µg/m3 lower on days that street washing was implemented, corresponding to a 15% reduction of road dust mass contribution relative to days when the road was left untreated.

In Finland, pressurized washing lowered PM10 emission levels by 15-60% (Kupiainen et al., 2011[100]). The effect was found to be highest immediately after treatment and dependent on the water pressure, the volume of water used, and the orientation of the nozzles in the pressure washer. The study found that the effectiveness of street washing is not only dependent on the characteristics of the cleaning equipment and washing method, but also on the frequency of cleaning and the amount of road dust, leading to a recommendation that street washing be considered where street dust levels are high and constitute the dominant source of PM10. Street washing functionalities can be incorporated into street sweepers or manually applied by hoses. An interesting case is that of some Korean cities, where some municipalities have employed self-cleaning road systems to tackle road dust emissions. The systems are generally operated twice a day during the summer. In the event of a heat wave, the system operates three times a day. The self-cleaning road system in Daegu has led to an estimated 5-15% drop in PM10 emissions (Kim, Jung and Kim, 2014[101]).

Street washing was found to reduce the road dust mobility by more than 90% in Spain and 60%-80% in Germany depending on the particle size and methodology applied (Amato et al., 2009[95]). The water volumes required for effective street sweeping may vary according to road dust loading. Although little investigated, an application rate of at least 1 litre per square meter is generally recommended. Road dust emissions levels have been shown to recover to 99% of pre-washing levels after 24 hours in Spain and 72 hours in the Netherlands (Amato et al., 2012b), as the moistening effect of water is more important than actual particle removal (wash off) in determining emissions levels. Given that peak emission period tends to be between 7 and 9am, the available evidence indicates that road washing will be most effective if performed in the early morning hours (4-5 AM) (AIRUSE, 2016[102]). Non-drinking water should ideally be used for street washing purposes, and possible trade-offs may exist regarding increased pollutants in sewage systems and natural waterways.

Within the AIRUSE LIFE project, street washing was also tested on industrial roads (Amato et al., 2016[103]). The interest in industrial sites was related to the higher PM emissions due to high road dust loadings. The paved road was located within the ceramic industrial cluster of L'Alcora (Spain). Road dust loadings (below 10 µm) were within 20-40 mg/m2, much above the general range in European cities of 1-6 mg/m2. Given such high road dust loadings, street washing was intensive (27 L/m2 phreatic water flow was used). Daily mean PM10 concentrations decreased by 18.5% on the day of the cleaning. However, this reduction was short lived, being reduced to only 2.2% on the day after cleaning. The main PM10 decrease occurred from 7 to 11 AM, which corresponds to 5 h after the start of cleaning activities. Although short lived, this decrease is sufficient to produce a significant reduction in daily mean PM10 concentrations (Amato et al., 2016[103]).

Box 4.4. The Barcelona street washing case

In Barcelona's city centre, the effect of a combined cleaning procedure consisting of vacuum-assisted broom sweeping and manual washing was tested. Averaging the daily mean concentration of PM10 over the 24 h following each cleaning event and on dry days, a mean decrease of 3.7-4.9 mg/m3 (7-10% of curbside concentrations) was found (Amato et al., 2014). Based on these results, the City Council implemented specific measures for improved road washing in the Air quality Plan 2015-2018. The measures include:

- Improve the state of road pavements
- Wash at 5 AM, in order to maximize effectiveness
- Wash the entire width of the road rather than the outer lanes only
- Prioritize high traffic roads
- Increase the frequency of washing during PM pollution episodes.
- Avoid the use of blowers for road cleaning

Source: City of Barcelona (2019[104])

Ideally, washing should be performed whenever road dust emissions are expected to be particularly high. The frequency of washing should therefore be optimized according to (1) local resources, (2) rain forecast (washing will be less effective within 24 h of rainfall), and (3) the episodic spikes in road dust (e.g. transported from the desert). If not all roads can be washed given resource constraints, a few roads should be selected for treatment to maximise the impact. The efficacy of road washing depends highly on several factors such as (1) climatic conditions, (2) road dust loadings, (3) frequency of washing, (4) road surface material, (5) portion and length of the road that is washed, and (6) relevance of other sources of PM.

Another set of measures are those aimed at reducing dust deposition, i.e. road dust loading. Trucks carrying dusty loads could be required to cover these loads, and trucks exiting building sites could be required to have their wheels and undercarriages washed. Reducing the amount of unpaved roads in urban areas and adopting the best available technologies to reduce dust generation during building activities, e.g. stockpiles, handling, loading/unloading) could also reduce road dust (EU, 2003[105]; Holman, 2014[106]; Regional Government of Styria Austria, 2006[107]; Environment Canada, 2005[108]; Swiss Agency for the Environment, 2004[109]; Greater London Authority, 2006[110]).

4.2. New strategies to target non-exhaust emissions

Stringent fuel efficiency regulations have succeeded in significantly reducing the vehicle exhaust PM emissions in recent decades. The evidence presented in Section 3 shows, however, that one of the most promising technological solutions for reducing exhaust emissions – namely the widespread penetration of electric vehicles – will not yield significant reductions in non-exhaust emissions. Given that the vast majority of all road transport PM emissions will come from non-exhaust sources in the coming years, similar attention should now be given to non-exhaust PM emissions.

Although the complete elimination of non-exhaust PM emissions is probably not realistic in the foreseeable future given that friction processes are inherent in acceleration and deceleration, the reduction of these emissions and the social costs associated with them is possible. PM from non-exhaust sources can be reduced by decreasing vehicle-kilometres travelled and lowering the amount of non-exhaust PM emitted per kilometre driven. A robust understanding of emission factors, their drivers, and the effectiveness of measures to reduce them will be necessary before being able to comprehensively assess the costs and benefits of various policy options. The analysis that follows is applied from a partial equilibrium perspective, as it does not consider potential trade-offs between reducing non-exhaust emissions and impacts on other environmentally-related outcomes.

This section reviews available policy options that can be used to reduce non-exhaust emissions. Policy instruments for addressing other environmental externalities include regulatory instruments, market-based instruments, as well as soft measures to encourage voluntary action on the part of producers and consumers. Section 4.2.1 discusses the relevance of an optimally-designed market-based instrument in the form of a charge designed to internalise the social costs associated with non-exhaust emissions. Section 4.2.2. identifies measures that can improve non-exhaust emission factors. Caveats and further considerations with respect to these policy options are also presented.

4.2.1. Reducing vehicle kilometres travelled

As reviewed in Section 4.1.2, a number of measures can be taken to reduce vehicle-kilometres travelled. To date, however, these measures have not targeted the reduction of non-exhaust emissions specifically, instead focusing on other negative externalities such as congestion and greenhouse gas emissions. Given the evidence presented in Sections 2 and 3, and in view of economic theory regarding the efficient internalisation of the costs of negative externalities, this section presents a framework for the design of a market-based instrument that addresses the negative externalities arising from non-exhaust emissions specifically.

The adverse impacts to human health entailed by non-exhaust emissions constitute the most significant negative externality associated with their generation and the main reason why policies targeting these emissions are needed. As these impacts can be considered a form of market failure, pricing instruments designed to internalise these external costs are an efficient, i.e. cost-effective, strategy to reduce emissions to optimal levels. At these levels, the marginal social benefit of an additional unit of emissions is equal to the marginal social cost of the damages associated with it (van Dender, 2019[111]; Parry et al., 2014[112]).[3] Importantly, economic efficiency only achieved if the charge targets the correct source of the negative externality in question and accurately reflects environmental damages. Equity issues stemming from the potential regressive distributional effects of the charge can be addressed through targeted compensatory mechanisms using part of the resulting fiscal dividends (Parry et al., 2014[112]).

Although non-exhaust emissions are primarily proportional to distance travelled, rather than fuel used, they also depend on factors other than distance travelled, namely vehicle weight, size, acceleration/deceleration speeds, road moisture, and road dust loading, as established in Section 2. The external cost per unit of non-exhaust emissions, moreover, depends on the number of people who are exposed to the PM caused by them. As such, a purely distance-based charge does not take into account the fact that the damages caused by non-exhaust emissions vary according to two important determinants of the magnitude of external costs, namely the characteristics of the vehicle, the area in which it travels, and the road surfaces.

To more accurately reflect these variations in marginal damages, therefore, charges designed to address the externalities from non-exhaust emissions should be differentiated according to the determinants of the marginal social costs they incur. A charge on non-exhaust emissions from road transport activity should drive both improvements in emission factors, as well as reductions in vehicle-kilometres driven (Parry et al., 2014[112]; van Dender, 2019[111]). An efficient non-exhaust emissions charge will be equal to the emission factor of a vehicle (g/vkm) times the marginal social cost of these emissions (EUR/g):

$$C_{ia} = f(E_i\,, D_a) \tag{1}$$

where C_{ia} is the per-kilometre charge for vehicle *i* in area a, E_i is the non-exhaust emission factor of vehicle i in grammes per kilometre, and D_a is the marginal damage cost of non-exhaust emissions per gramme in area a. Importantly, this charge differs from a flat per-kilometre charge because it is differentiated with respect to the determinants of the marginal external cost of non-exhaust emissions, namely by the characteristics of the vehicle and the area in which it operates.[4] The emission factor E is indexed by the vehicle *i* and the marginal damage is indexed by area *a*:

$$E_i = f(w_i, b_i, t_i) \tag{2}$$

$$D_a = f(p_a, c_a) \tag{3}$$

Where the emission factors are determined by w_i, which represents the vehicle weight, b_i the relevant brake characteristics, and t_i the tyre characteristics.[5] Marginal damages D_a are a function of p_a, the population density of an area and c_a, the level of congestion. Because the purpose is to take into account population exposure, the definition of each area would ideally be informed by, for example, measures of population density, the general proximity of buildings to roadways, and travel via soft modes that is undertaken (where mode shares are not available, the extent of soft-mode infrastructure available could be used as a proxy). Following EU standards for CO2 emissions, for example, the charge could vary according to whether travel took place in urban, suburban, or rural areas, or on motorways. If marginal damages are found to depend on the ambient air quality, this could be reflected by a scale parameter that increases the charge for all vehicles when air quality is already poor.

This type of charge combines the spatial and temporal specificities of a congestion charge with the distance-based characteristic of a flat per-kilometre charge in order to accurately reflect variations in the marginal social damages entailed by non-exhaust emissions. The specific form that these functions should take will rely on more extensive evidence than currently exists regarding the drivers and impacts of non-exhaust emissions. Their design will notably require the establishment of reliable estimates of marginal damages based on emission factors and health impacts (see Section 3.3). To the extent that the implementation of this instrument makes vehicles with lower non-exhaust emission factors more attractive to consumers, it theoretically also incentivises manufacturers to produce vehicles and vehicle parts (mainly tyres and brakes) with lower emission factors if these emissions factors are made known to prospective consumers.

The fact that non-exhaust PM emissions can also contain larger particles than exhaust PM emissions (i.e. they settle more rapidly, thus affecting a smaller area), means that the need to prioritise densely populated areas is even greater for measures targeting non-exhaust emissions than for measures targeting exhaust emissions. The design of a pricing instrument targeting non-exhaust emissions should involve context-specific evidence on emission factors, health impacts, as well as any insights arising from the previous implementation of distance-based charges. The implementation of such an instrument would realistically need to be automated, for example based on GPS data (as suggested in van Dender (2019[111])). Additional considerations that should be taken into consideration in the design of such a measure are provided in Section 4.3.3.

Goodkind et al. (2019[113]) estimate marginal damages of primary and secondary PM2.5 from exhaust emissions, as well as from road dust resuspension, but not from direct wear sources (Table 4.4). They also assume no difference in health impacts across different PM2.5 species, and value mortality only. Fann, Baker, and Fulcher (2012[114]) also estimate the benefits of reducing primary PM as well as secondary PM from exhaust sources, reporting benefits of 2015 USD 400,000, 81,000, and 21,000 per tonne of PM avoided.[6] To date, no studies have estimated the marginal damages associated with exposure to non-exhaust emissions specifically.

Table 4.4. Median marginal damages of PM2.5 emissions from road transport

(thousand 2011 USD per tonne)

	Light petrol vehicles	Heavy petrol vehicles	Light diesel vehicles	Heavy diesel vehicles	Dust: paved roads	Dust: unpaved roads
Primary PM2.5	122.44	88.97	116.53	83.97	64.43	34.72
Secondary PM2.5						
VOC	7.18	6.25	7.49	5.96	0	0
NH3	73.58	61.20	68.44	50.69	0	0
SO2	24.32	22.61	24.4	22.54	0	0
NOx	12.55	11.72	13.37	12.54	0	0

Source: Goodkind et al. (2019[113])

4.2.2. Improving emission factors

Given the importance of vehicle weight as a determinant of non-exhaust emissions, vehicle lightweighting should be considered an important regulatory tool for addressing the non-exhaust particulate matter emitted by both ICEVs and BEVs. Reducing vehicle weight will lead to less brake, tyre, and perhaps also road wear. In addition to the maximum allowed weight for selected roads, other measures to reduce HDV overloading include improved enforcement mechanisms and higher fines (Žnidarič, 2015[16]). Overloading of goods vehicles has been estimated to cost the UK over GBP 50 million a year (2003 PPP) through the additional wear and tear that it creates on roads and bridges. Overloaded drive axles are the biggest single cause of excessive wear and tear on roads in the country (UK Department for Transport, 2003[115]).

In a similar way as increasing stringency has led to improved fuel efficiency, regulations on the materials that make up brake pads, brake rotor discs, and tyres have led to reductions in the toxicity of PM emissions. Regulations regarding not only the toxicity of brake and tyre materials, but also their durability would more directly target the amount of particulate matter emitted from tyres and non-regenerative braking systems. Nevertheless, regulations affecting brake and tyre composition such as asbestos bans, REACH, CLP, and restrictions on the use of trace elements and heavy metals should continue to be adopted in countries where they are not yet implemented. These regulations should also become increasingly stringent both in terms of the minimum concentration thresholds they require and in the number of toxic components they consider. Particular attention should be given to heavy metals and organic compounds of known toxicity. Given the nascency of the evidence regarding the health impacts of non-exhaust emissions, efforts should also be undertaken to make regulations more easily adaptable to emerging scientific evidence.

More generally, continued technological development, supported by incentives for innovation to further reduce emission factors, will contribute to reducing non-exhaust emissions. Policies favouring vehicle lightweighting should be put in place. Driver assistant systems are another existing technology that can influence driver behaviour to be more efficient so as to reduce non-exhaust emissions from brake and tyre wear. Sophisticated DAS can include sensors for rotor temperature and tyre pressure, as they are important parameters for brake and tyre emissions, respectively. The more these systems reach automated driving, taking the human driver out of the loop, the more effective they will be in reducing non-exhaust emissions.

The cost-effectiveness of road management measures, such as dust binding and street washing, should be evaluated on a case-by-case basis. The use of dust suppressants, for example, should involve a consideration of potential environmental trade-offs, including damage to vegetation and human health and contamination of soil and ground water. Other potential side-effects include reducing road friction and corrosion to highway infrastructure such as bridges and the road surfaces. In areas where water is scarce, street washing may not be cost-effective.

Eco-labelling of brakes and tyres can be a useful element of regulations to increase consumer awareness about the negative impacts of non-exhaust emissions. As a first step, eco-labelling can reflect the content of toxic components, as is already practiced in the US states of California and Washington. Ultimately, eco-labelling could be implemented in the same way as CO2 emissions labelling, which is mandatory in the EU, although this would rely on the development of a standardized measurement protocol for non-exhaust emissions.

4.3. Knowledge gaps and an agenda for future research

Evident challenges exist with respect to developing policies to internalise the externalities of non-exhaust emissions. These include measuring non-exhaust emissions, measuring and valuing exposure to non-exhaust emissions, and designing and implementing the policy. For example, the implementation of a charge on non-exhaust emissions in road transport is predicated on being able to quantify the marginal damages associated with these emissions as well as establishing a robust understanding of the quantitative nature of the relationship between the various drivers of non-exhaust emissions (e.g. vehicle weight) on which emission factors (and thus the charge amount) would be based. As evident in Section 2, considerable uncertainty remains with respect to these relationships.

4.3.1. Measuring and sourcing non-exhaust emissions

First, standardized methodologies to measure brake, tyre and road wear emissions will need to be developed in order to design, implement, and enforce the policies presented in the previous section. There is currently a lack of such standardized measurement methods that use reliable sampling equipment and have the capacity to replicate real-world conditions. This has a direct influence on the quality and harmonization of emissions inventorying, air quality modelling and projections, since the values available in literature are obtained using a wide range of instrumentation, testing and sampling/monitoring procedures. As a result, there is a need for harmonization and consistency in emissions reporting, in the representativeness of emission factors, and the inclusion of resuspension.

For brake wear emissions, the LOWBRASYS project has provided suggestions regarding the development of a suitable methodology for sampling and measuring brake wear particles as well as recommendations for the development of a representative of real-world conditions braking cycle, concluding that the developed 3h-LACT can be used for standardization and thus regulatory purposes as it fulfils the requirements for a real-world representative braking cycle (Kousoulidou, Perricone and Grigoratos, 2019[36]). More recently, an important step-forward towards a commonly accepted methodology has been made by the UNECE informal working group on the Particle Measurement Programme (UNECE, 2019[116]), which has developed a novel, publicly available WLTP-based real-world brake wear cycle[7] (Mathissen et al., 2018[117]). Similar initiatives should be taken for tyre wear and road surface (wear and resuspension) emissions. For brake wear, it is also important to conduct research on the benefits and drawbacks of different combinations of brake pads and rotors.

For road and tyre wear emissions, the influence of pavement type and structure remains largely unknown. For tyre wear, it is necessary to conduct research on the benefits and drawbacks of different combinations of road and tyre materials. Most of the available literature is based on studded tyres conditions; thus more research is needed in a standard tyres context. More research is also required to quantify the effect of low-rolling resistant tyres on tyre wear. Although durable rocks (e.g. porphyry) seem to be a logical choice in road pavement material, this material might be rare and its use not economically defensible in areas where road wear is of low priority. Road pavement characterization should accompany air quality studies dedicated to assess the impact of road wear. The impact of surface texture and state should be further investigated since a larger grain size and surface porosity appear to be associated with lower road wear emissions. The possible impact of alternative fillers (e.g. rubber and cement) and aggregates (e.g. furnace slag) should also be explored. A standardized test could conceivably assess the PM emission properties of pavement samples. Such a method could help to understand which pavement construction types and which pavement and rock aggregate properties are the most important determinants of PM emissions. This could facilitate the choice of pavements in environments where pavement wear is an important source of emissions. The Prall and the Tröger test methods could be useful methods in this regard, as suggested by Snilsberg (2008[118]), but they will require further development for the specific purpose of particle emission investigations.

For road dust resuspension, the local environment is even more important than for road wear, since both road dust loading and meteorology are important drivers of PM emissions from road dust. As current emission factors from resuspension may include wear emissions, new methods to avoid double counting should be developed. However, given the dominant role of resuspension among non-exhaust emission factors, the magnitude of double counting would probably be low relatively to the magnitude of error that is entailed by discarding resuspension. Local measurement initiatives that couple road dust loadings, mobile vehicle measurements and ambient air measurements, would significantly advance this effort. In this way, different measurement techniques can be used together to pursue a better understanding of the relationship between direct wear and road dust resuspension rates and their contributions to ambient air pollution (Denby, Kupiainen and Gustafsson, 2018[119]). A number of measurement methodologies have been used to measure road dust loadings on an ad-hoc basis (e.g. wet and dry sampling, different cut-offs). Comparing these methods and developing a harmonized methodology is a prerequisite for considering road dust resuspension in emissions inventories.

The relationship between road dust emission factors and vehicle weight also warrants further study. Although the analysis presented here does not assume a qualitative functional form for the relationship between road dust emission factors and vehicle weight, a positive relationship between the two would imply that BEVs have greater road dust emission factors than ICEVs, making policy measures targeting non-exhaust emissions even more of a priority as EV uptake spreads. Another avenue for future work would be to explore the possible impact of flatter vehicle undercarriages on road dust emissions.

Further research will also be necessary in order to better separate the contributions from road dust resuspension, brake, and tyre and road wear, given that their relative toxicity for human health and the potential need for regulations to address them are likely to be different. In this sense, valuable information can be offered by size and time-resolved PM chemical characterization and particle size distribution measurement, as well as improved source apportionment tools. Road wear studies are still lacking in countries with non-studded tyres. This is particularly the case in Central and South America, Africa and the Middle East, where both non-exhaust emissions and urban population densities – and thus damages – are likely to be high.

Further research is also needed in order to identify and quantify the impact of vehicle, road, driver and environmental features on the non-exhaust emission potential. The influence of vehicle weight, size and speed on emission factors calls for detailed investigation. More research is also needed on how driving style impacts emission factors, which may point to significant opportunities to reduce emissions, especially enabled by technological developments in DAS. The influence of vehicle maintenance (e.g. tyre pressure, brake pad replacement) should also be investigated as a determinant of non-exhaust emissions. More experimental studies on PM emissions from EVs are also warranted. Currently, comparative studies with ICEVs rely on several assumptions on vehicle weight, the relative emissions reductions made possible by regenerative braking and low rolling resistance tyres. More research on the contribution to secondary PM from ICEVs is needed in order to properly assess their emissions relative to EVs.

4.3.2. Measuring and valuing the damages caused by non-exhaust emissions

With respect to estimates of the impact of non-exhaust emissions on health, more studies using source contributions as exposure indicators are needed, and use of state-of-the-art methods will reduce uncertainty in the use of source contribution data (effect modification, residual analysis, and multipollutant modelling). Although non-exhaust emissions have been associated with a number of short- and long-term health effects that are similar or higher in magnitude to those found for exhaust emissions, more epidemiological studies are nevertheless needed in order to carry out meta-analyses for different causes of mortality and morbidity. Although the effects of long-term exposure to non-exhaust PM have been reported in cohort studies, this evidence is still inconsistent, except for copper, iron (often associated with brake wear) and zinc (often associated with tyre wear), which have been strongly linked to mortality from cardiovascular causes.

From the toxicological point of view, although there are few in vivo toxicity studies focused specifically on non-exhaust sources, emerging data indicate that non-exhaust PM can be as hazardous as tailpipe PM depending on the nature of the health effect studied (Amato et al., 2014b). This is because the oxidative stress which is often related to transition metals and/or redox active organics is one of the main biological mechanisms behind the toxicity of PM. Other important factors to be investigated are the roles played by PM size and size distribution, particle number, composition (including coating and surface modifications), shape, surface area, surface chemistry, and charge and solubility/dispersability, on health impacts. The possibility of non-linearity of these health impacts with respect to ambient PM emissions should also be investigated, as this can inform the design of more sophisticated policy measures to reduce non-exhaust emissions.[8]

Most epidemiological studies use chemical elements as indicators of exposure, but since there are no unique tracers for non-exhaust emissions, source contributions should rather be used as exposure indicators (together with their uncertainty estimates), which would allow for a better separation of non-exhaust processes as well as a direct comparison with exhaust emissions. Estimates of the benefits of reducing PM emissions from road transport should be developed for non-exhaust emissions, as the health impacts could be different from the primary and secondary PM from exhaust emissions that have already been estimated.

4.3.3. Designing effective mitigation measures to reduce emission factors and vehicle kilometres driven

More research is also needed in order to advance an understanding of the most effective measures for reducing emission factors and vehicle kilometres driven. For brake wear emissions, more research could be undertaken to optimize the mechanical relationships between brake rotor, clamp and overall design of braking systems. Continued activity in testing novel brake pad and brake rotor disc composition could also prove fruitful. An important avenue for future research on road wear and road dust resuspension concerns the role of pavement texture and mineralogy under a variety of meteorological conditions (not only in countries with snow and ice).

Finally, a number of factors should be carefully considered in the design of pricing instruments aiming to internalise the external costs entailed by non-exhaust emissions. Complementary measures or provisions could address the possible distributional effects associated with a charge for non-exhaust emissions. Consideration should also be given to how the revenues of the charge will be spent, as this can influence the social acceptability of the measure. They could be spent on additional remedial measures to mitigate non-exhaust emissions (e.g. wetting the pavement), which will depend on local conditions and costs. Given that vehicle registration fees have been shown to have a strong effect on vehicle choice, incorporating vehicle weight and possibly brake/tyre material into these fees also has potential as a complementary intervention at the point of vehicle choice. It will also be necessary to anticipate, to the extent possible, the potential for unanticipated rebound and spillover effects. When combined with a pricing disincentive addressing non-exhaust emissions, measures that improve non-exhaust emission factors may be susceptible to rebound effects (Dimitropoulos, Oueslati and Sintek, 2018[120]). Rebound effects occur if lower emissions factors and thus a lower per-kilometre cost from a non-exhaust emissions charge would induce people to travel more than they otherwise would have.

The estimated comparison between BEV and ICEVs emissions is built on different state-of-the-art methods. Future studies would indeed benefit from an improved knowledge on several aspects which represent the main limitations of this study, including more accurate data on mean weight estimates for EVs and their internal combustion counterparts, more accurate (experimental) relationships between vehicle weight/size and non-exhaust emissions, experimental measurement of emission reduction due to regenerative braking, and more accurate estimates of the contribution of different fuel engines to secondary aerosols formation (inorganic and organic).

4.4. Implications for policy

Consensus exists in the scientific literature that non-exhaust emissions are becoming the dominant source of PM from road traffic and that PM, and particularly PM2.5, have nefarious effects on human health. Despite these demonstrated negative externalities, non-exhaust emissions have been only tangentially addressed by public policies to date. Given the magnitude of the aggregate social costs they entail, and the fact that the transition to electric vehicles will not lead to significant reductions in non-exhaust emissions, policymakers should invest resources in determining how to optimally reduce them via targeted policy instruments.

To the extent that developing policies to address non-exhaust emissions will rely on a well-developed understanding of their drivers, priority should be given to the development of a standardized measurement methodology for each of the processes that generate non-exhaust emissions. Some progress has been made in this regard for brake wear, but similar initiative remains to be taken for the measurement of tyre wear, road wear and road dust resuspension. With respect to road dust emissions, even when a commonly accepted methodology has been developed, regulation based on these emissions would be difficult to apply in practice given that emissions will depend heavily on local meteorological and environmental features.

An optimally designed non-exhaust emissions charge would theoretically incentivise the reduction of non-exhaust emissions by all relevant actors (e.g. consumers and manufacturers). No precedent currently exists for such a charge, in part because non-exhaust emissions and their social costs have gone largely unnoticed, but also because of a lack of robust evidence quantifying the relationships between various vehicle characteristics and their impact on emissions. Limitations with respect to the evidence required to design such a charge, as well as the practical challenges of implementing this charge mean that other measures will also be necessary in order to address these emissions.

Given the high proportion of non-exhaust emissions generated by tyre wear, priority should be placed on measures that seek to reduce PM emissions from this source in particular, namely vehicle lightweighting and regulations on tyre composition. To the extent that lighter vehicles also emit less PM emissions from brake wear, policies that more explicitly favour vehicle lightweighting can be a particularly effective tool for addressing non-exhaust emissions.[9] Such policies could include the expansion of weight-based charges. Investing in R&D to develop lighter materials (e.g. high-strength steel and aluminium alloy) will also advance this agenda.

Insofar as population exposure is greatest in urban areas and current congestion pricing schemes are an effective means of reducing motor vehicle traffic in these areas, another policy priority for addressing non-exhaust emissions is the more widespread use of congestion pricing in city centres. These pricing mechanisms could be further optimised to target non-exhaust emissions, for example, by finding ways to differentiate electronic congestion charges by vehicle weight and distance travelled, e.g. through the use of registration data and GPS technology.

Promising avenues for technological developments that stand to reduce emission factors are regenerative braking systems and brake dust collection devices. While already used for electric and hybrid vehicles, it is also worth exploring the degree to which RBS could be easily incorporated into ICEVs given that they will continue to dominate vehicle fleets in the coming years. Preliminary indications suggest that brake dust collecting devices also have mitigation potential, but they have not yet been introduced into the market at the level of original equipment manufacturers or as aftermarket solutions.

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[1] One solution of the former type would involve changing urban travel altogether by switching to technology that does not rely on friction for acceleration and deceleration (e.g. propulsion via

magnets, as is used by Maglev trains). This report does not explore such a fundamental shift, as it is not a realistic possibility for micro-level urban passenger travel in the near term.

[2] For more information, see www.ertico.com.

[3] Although greater distance travelled implies greater fuel use, the opposite is not necessarily true, e.g. the case of an idling car.

[4] Based on the evidence reviewed in Section 2, other features that should be taken into account include driving style, pavement type, road moisture and road dust loading. Practically speaking, however, it would be difficult to design a road charge based on these characteristics.

[5] Including brake and tyre characteristics as determinants of a vehicle's emission factor implies the development of standardised methodologies for the measurement of non-exhaust emissions from these sources. Even though non-exhaust emissions have been found to increase with vehicle weight, more research is needed to establish a robust quantitative relationship between vehicle weight and non-exhaust emissions.

[6] These benefits reflect the value of the avoided cases of premature death and illnesses avoided, quantified through a combination of willingness to pay and cost of illness estimates.

[7] Available at https://data.mendeley.com/datasets/dkp376g3m8/1.

[8] The marginal external cost of congestion increases non-linearly with traffic load: when roads are more congested, the cost of each additional car in circulation becomes greater. The extent to which the relationship between exposure to an increased concentration of PM emissions varies by ambient air quality should also be explored to investigate whether the negative health impacts of non-exhaust emissions may follow a similar pattern.

[9] Vehicle lightweighting, moreover, also reduces greenhouse gas emissions (ITF, 2017[15]).

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