Non-Exhaust PM Emissions from Battery Electric Vehicles (BEVs)

– does the argument against electric vehicles stack up?

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1. Introduction

Non-exhaust emissions of particulate matter (PM) from motor vehicles is becoming a hot topic at the moment. In particular the expected impact of battery electric vehicles (BEVs) on non-exhaust PM emissions.

There appears to be an emerging view that **battery electric vehicles** (BEVs) will not contribute to a reduction, and perhaps even increase emissions of non-exhaust PM from the current on-road fleet of (mostly) fossil-fuelled **internal combustion engine vehicles** (ICEVs).

However, the arguments and assumptions used by researchers differ substantially, as will be discussed in this study. More importantly, the assumptions are often simplified and have limited or even no empirical basis, which brings into question the accuracy and robustness of the conclusions.

This study will examine and unpack the arguments underpinning statements made regarding BEVs and non-exhaust emissions, critically review the validity of assumptions made, and conclude with a probabilistic analysis to estimate the likely impact of BEVs on non-exhaust PM emissions. It is noted that the study focus is on PM mass emissions, which aligns with current PM air quality criteria, and not particle number (PN). Further research into PN emissions (exhaust and non-exhaust) is recommended.

Particulate Matter

Airborne particulate matter (PM) encompasses a wide particle size range, from a diameter of a few nanometers to around 100 micrometer. PM is is typically classified as PM_{10} and $PM_{2.5}$, which represents particles with a (aerodynamic) diameter of less than 10 μm and 2.5 μm, respectively.

Vehicles emit PM through their exhaust pipes as well as through other non-exhaust (abrasive) mechanisms, such as tyre wear, brake wear, road surface wear, clutch wear, resuspension of road dust, and even corrosion of vehicles and street furniture.[1,2]

Traffic intensity is one of the most important determinants of ambient man-made PM concentrations, and people living in cities and near major traffic routes are particularly affected by high PM concentration levels. The contribution of road traffic to local PM concentration levels can be up to 65% on busy streets.[19]

2. Vehicle emissions in a nutshell – Battery Electric Vehicles (BEVs) versus Internal Combustion Engine Vehicles (ICEVs)

BEVs have several clear benefits over ICEVs.

Moving from ICEVs to BEVs for light passenger vehicles will have significant benefits with respect to energy efficiency and reduction of greenhouse gas emissions from the transport sector. Compared to conventional fossil-fuelled and hydrogen (passenger) vehicles, BEVs will be the most robust option with regard to moving to a zero greenhouse gas (GHG) emission road transport system. Compared to ICEVs they can enable (V2G) and have the potential for significant to very deep (98%) reductions in greenhouse gas emissions from the on-road fleet, where the size of the reduction depends on the extent of renewable energy generation in the electricity system.[2,3] Even for the largely coal-based electricity generation in Australia, BEVs will achieve significant GHG emission reductions of about 40% (‘Current Australia’), to deep cuts of about 75% (‘More Sustainable Australia’), to very deep cuts of 98% (‘Decarbonised Australia’).[3]

Ambient air pollution is associated with a wide range of public health effects, ranging from minor respiratory tract irritation to increased mortality. The close proximity of motor vehicles to the general population makes fossil-fuelled motor vehicles (ICEVs) one of the most important sources of local air pollution, particularly in urban areas. Although emissions per kilometre of driving have reduced over time, ICEVs still emit substantial amounts of air pollution, simply because of the increasing number of vehicles on the road.

BEVs, when substituted for ICEVs, improve local air quality, particularly in urban areas where population and associated transport movements are concentrated. This is because BEVs are powered by an electric motor, which uses electricity stored in a battery and is plugged-in to recharge. BEVs do not have exhaust pipes or fuel tanks/systems, which means BEVs do not produce exhaust emissions or evaporative (fuel vapour) emissions (fuel tank, fuel lines, leakage, etc.). Hence, BEVs are often referred to as 'zero-emission' vehicles. However, the level of improvement in local air quality due to fleet electrification depends on how (fuel mix) and where (e.g. distance of power stations to urban areas) electricity is generated, as will be discussed later.

The only local air pollutants that *all* on-road vehicles produce are non-exhaust particulate matter emissions due to brake wear (wear of brake lining and brake disc/drum)[1], tyre/road wear (wear due to frictional contact between the road surface and tyre tread) and resuspended road dust (either vehicle induced or wind-induced).

On this aspect there appears to be significant variability in the scientific literature and points of contention.[2] Some publications conclude that BEVs have similar non-exhaust PM emissions to fossil-fuelled ICEVs.[4-6] Others state that electric vehicles increase non-exhaust PM emissions.[7] It has also been stated that BEVs slightly increase non-exhaust PM_{10} emissions, but reduce $PM_{2.5}$ emissions.[8] In contrast, others conclude that, on balance, all non-exhaust PM emissions by BEVs are reduced.[9] So there is substantial variability in the conclusions within publications regarding non-exhaust PM impacts of BEVs as compared with ICEVs.

3. Exhaust versus non-exhaust emissions

Exhaust and non-exhaust PM emissions are quite different with respect to particle size distribution and chemical characterisation. Exhaust PM emissions are mainly made up of $PM_{2.5}$ and contain a variety of hydrocarbons, which can contribute to respiratory disease and may lead to cancer. For instance, diesel exhaust is classified as a carcinogen by the International Agency for Research on Cancer (IARC).[10] This was based on sufficient evidence that exposure to diesel exhaust causes lung cancer.[2] Non-exhaust PM emissions tend to contain mostly PM_{10}.

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[1] There are two main friction brake system configurations in current use: disc brakes, in which flat brake pads are forced against a rotating metal disc (usually cast iron or reinforced aluminium), and drum brakes, in which curved brake shoes are forced against the inner surface of a rotating cylinder. Cars are usually equipped with front disc brakes and either rear disc or drum brakes, and many modern vehicles are equipped with anti-lock braking systems (ABS). Commercial vehicles tend to be fitted with drum brakes, although disc brakes are being introduced by some manufacturers.

[2] People are exposed not only to motor vehicle exhaust emissions (cars, trucks) but also to exhaust emissions from other diesel engines, including from other modes of transport (e.g. diesel trains and ships) and from power generators.

Non-exhaust PM emissions contain significant amounts of heavy metals such as zinc (tyres), iron (brakes) and copper (brakes/tyres), antimony (brakes), amongst others.[11-13] In fact, brake and tyre wear have been identified as an important source of trace metals in the urban environment, where most of the population live and work.[14,15] Other research suggests that tyre abrasion is a major contributor to both black carbon (BC) and polycyclic aromatic hydrocarbons (PAHs) in soils.[12,16,17]

Toxicological studies have reported on associations between non-exhaust PM from road traffic and negative health effects, such as impacts on autonomic control of heart rhythm, lung-inflammation and DNA damage.[17-21]

4. The importance of non-exhaust emissions

4.1 National scale

The significance of exhaust and non-exhaust PM emissions differs between countries. It reflects the on-road fleet in terms of local vehicle emission standards, fuel mix, maintenance practice, etc.

Motor vehicle emission inventories (MVEIs) generally suggest that non-exhaust PM generally dominates total PM emissions (i.e. exhaust + non-exhaust). Overseas studies report that non-exhaust PM will contribute as much as 50% to 90% to total PM emissions from road traffic, due to increasingly stringent exhaust PM emission standards.[20,22,23] For the purposes of this study, TER used the Australian Fleet Model (AFM) to create an input file for COPERT Australia, reflecting the Australian on-road fleet for the base year 2020. COPERT Australia was then used to create a national Australian MVEI for 2020. The MVEI estimates that the proportion of non-exhaust PM emissions (i.e. brake, tyre, road wear) to total national PM emissions from motor vehicles is about 70% ($PM_{2.5}$) and 80% (PM_{10}) for the Australian fleet in 2020. This result is well within contributions reported for other countries.

4.2 Local Scale

Significant uncertainty is associated with non-exhaust PM emissions at a local scale. For instance, a source apportionment study in a tunnel [15] found that only about 20% of PM concentrations could be linked to brake and road wear, whereas 30% were contributed to both exhaust emissions and resuspension. Indeed, source apportionment of PM emissions is often difficult due to the problem of double counting and the lack of unambiguous tracers.[22] A number of studies have found approximately equal contributions for exhaust and non-exhaust PM from traffic sources [24], with up to 90% for non-exhaust in 'special cases', i.e. northern EU countries that use road traction sand and studded tyres[3] in winter.[11,25,26]

[3] Winter tyres may include e.g. metal studs to enhance their friction under icy conditions. Friction tyres do not have studs but have a special tyre tread design and material, for example soft rubber material and more lamellae than summer tyres. Most used tyre models in the cold climate countries (Sweden, Norway, Finland, Canada, Russia, USA) differ from the ones used in e.g. central European countries in the respect that the Nordic designs emphasise more snow and ice grip, whereas the central European designs performance against aquaplaning.[26]

The variability at a local scale is not surprising given the many factors that influence real-world non-exhaust PM emission levels. These include vehicle speed, vehicle weight, severity of braking, vehicle conditions, maintenance history, road surface (concrete, asphalt, unpaved), tyre characteristics, etc. (see box on next page).

A comprehensive review of almost 100 research studies concluded that road dust (road wear and resuspended dust combined) had, on average, an almost equivalent contribution as vehicle exhaust to PM_{10} concentrations, both contributing about 20%.[27] Brake wear and tyre wear had lower average contributions to PM_{10}, i.e. 7% and 4%, respectively. For $PM_{2.5}$, the mean contributions were 24% (vehicle exhaust), 11% (road dust), 9% (brake wear) and 2% (tyre wear). This study concludes that, regardless of the environment studied, non-exhaust vehicle emissions are already at least as important as vehicle exhaust for PM_{10}, and almost as important for $PM_{2.5}$.

Factors influencing non-exhaust PM

There are many factors that influence non-exhaust emissions.

Most tyre rubber is lost during acceleration, braking, and cornering, so the amount of rubber lost will therefore tend to be greatest near busy junctions and on bends.[28]

As brake wear only coours under braking, the highest concentrations of brake wear particles should be observed near busy junctions, traffic lights, pedestrian crossings and corners. Then again, particles may also be released from the brake mechanism or wheel housing sometime after the primary emission event.[28] In addition, the range of brake lining materials in use, and variations in brake lining formulations has a major influence on emission levels.[11]

In essence, non-exhaust PM emission processes are complex. For instance, tyre wear can be simulated for individual tyres with complex mathematical models including a range of predictor variables such as vertical load, width of the tyre, angular velocity, damping coefficient, wear coefficients, slip ratio elastic modulus of rubber, and several others.[29]

4.3 Conclusion on importance

It is clear from international research that non-exhaust PM emissions are important for current on-road fleets around the world and in Australia. They can contribute significantly, or even dominate local PM concentrations, but the contribution is highly variable and critically location dependent.

5. BEVs and non-exhaust emissions

A review of international literature shows that various studies make statements regarding the relative impact of BEVs on non-exhaust PM emissions when compared with fossil-fuelled ICEVs (refer to section 2). The arguments and assumption used by researchers often differ substantially. More importantly, they appear too simplified and have often limited or even no empirical basis, which brings into question the accuracy and robustness of the conclusions. The main points of contention are discussed in this section.

5.1 BEVs are heavier than ICEVs

The weight of a vehicle is generally accepted to be an important factor in relation to non-exhaust PM emissions. Heavier vehicles generate higher loads on tyres and road surface, leading to increased wear. Similarly, heavier vehicles have more inertia, causing more friction between brake pads and wheels leading to more brake wear. So vehicle weight is important with regard to non-exhaust PM emissions. The shape of these relationship between wear and weight is, however, not universally agreed, and both linear and non-linear impact of weight on wear have been used to estimate non-exhaust PM emissions, leading to different outcomes.

To compare non-exhaust PM emissions between ICEVs and BEVs, a (mean) difference in weight needs to be determined. This is, however, not as straight-forward as it sounds, because BEVs do not fit well into conventional ICEV vehicle classification schemes. One approach is to allocate BEVs to specific ICEV market segments and compute the difference in mean weight. Another approach is to compare only BEVs that have an equivalent non-electric ICEV version (e.g. VW Golf vs. VW e-Golf).

The issue is that, in either case, allocation is quite arbitrary, so the comparison may or may not be adequate or accurate. The main underlying assumption is that vehicle purchasing behaviour does not change when switching from and ICEV to a BEV. This seems unlikely. Purchase price and performance characteristics (e.g. rated power, fuel/energy use) will largely guide purchase decisions. It is thus quite possible that ICEV owners would buy a BEV in a different class. This would then also lead to a different impact on non-exhaust PM emissions.

The general conclusion from overseas studies is that BEVs are typically about 20-25% heavier (about 200-300 kg) than ICEVs, due mainly to the weight of the battery. As a consequence, BEVs have been estimated to produce a proportional increase in road and tyre wear [30], but less brake wear due to regenerative braking, as will discussed in the next section. Other studies have used a smaller weight penalty for tyre wear of 10%, and have assumed no measurable impact on road wear.[9]

It is noted that most BEVs come equipped with low rolling resistance tyres to save energy and have higher tyre pressures, which may lead to lower wear and associated PM emissions.[1,2,31]

The question is: does this assumption equally apply to Australia? In order to answer this, analysis of vehicles sales data is required. TER (2019) created an Australian car/SUV database using fuzzy matching to stitch VFACTS and Green Vehicle Guide databases together.[32]

Figure 1 and Tables 1 and 2 show an analysis of vehicle weight information in national sales data VFACTS [4]. The VFACTS data provides detailed vehicle sales information for specific vehicles by year of sale (retail).[33] Detailed vehicle information is also provided, including vehicle class (car, SUV, commercial), fuel type (petrol, diesel, LPG, electric), drive train (ICEV, Hybrid, PHEV, EV), brand, model, engine capacity, tare weight (TW) and gross vehicle weight (GVW). The data were used in a previous TER study [32] to compute sales-weighted 'on-road' weight. On-road weight was estimated as (TW + GVW)/2.

The analysis shows that both Australian diesel cars and SUVs are significantly heavier than their petrol counterparts, and that this weight difference is increasing, up to 500-600 kg (about 30-40%) in recent years. In contrast, Australian BEVs in the passenger car class have similar weights to their petrol counter parts. Australian Diesel cars are on average about 25-30% heavier than Australian BEVs. For SUVs, BEV data is limited reflecting limited sales of electric vehicles in this segment. Figure 1 shows that there is only one data point representing one SUV BEV in one year (2018) and the weight is similar to the diesel SUVs.

Table 1 – Average weights for Australian passenger cars (sales-weighted).

Year of Sale	ICEV petrol (kg)	ICEV diesel (kg)	BEV (kg)	Difference (BEV - A)/A	
				A = ICEV-P	A = ICEV-D
2010	1579	1747	1230	-22%	-30%
2011	1566	1788	1434	-8%	-20%
2012	1558	1811	1467	-6%	-19%
2013	1555	1832	1757	+13%	-4%
2014	1569	1845	1738	+11%	-6%
2015	1578	1912	1666	+6%	-13%
2016	1595	2000	1639	+3%	-18%
2017	1597	2068	1486	-7%	-28%
2018	1573	2153	1620	+3%	-25%

Table 2 – Average weights for Australian SUVs (sales-weighted).

Year of Sale	ICEV petrol (kg)	ICEV diesel (kg)	BEV (kg)	Difference (BEV - A)/A	
				A = ICEV-P	A = ICEV-D
2010	1950	2356	-	-	-
2011	1922	2354	-	-	-
2012	1896	2341	-	-	-
2013	1883	2304	-	-	-
2014	1878	2343	-	-	-
2015	1841	2345	-	-	-
2016	1818	2366	-	-	-
2017	1820	2380	-	-	-
2018	1824	2421	2402	32%	-1%

[4] © Federal Chamber of Automotive Industries (2019).
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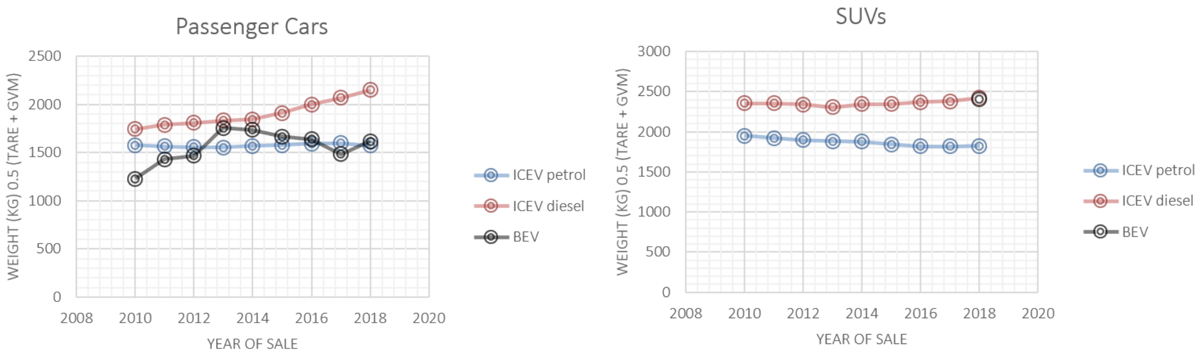

Figure 1 – Average sales-weighted weights for Australian passenger vehicles.

The analysis shows that for the Australian fleet, the weight difference for BEVs currently available in the Australian market, is not an issue as they have similar weight to petrol cars, and are on average substantially lighter than diesel cars. This means that, based on weight, diesel cars and SUVs appear to be the most important source of non-exhaust PM emissions in the on-road passenger fleet, not BEVs.

Fossil-fuelled passenger cars have become heavier and more powerful over time, both in Australia and overseas.[32,34] A 'SUV boom' is observed around the world. This trend of buying heavier vehicles is particularly strong for diesel cars, as is evident in Figure 1, and also reported overseas.[34]

In terms of weight difference, the focus on BEVs is therefore not justified, and should rather focus on the (increasingly) heavier diesel cars and SUVs in the Australian fleet, especially given the sustained growth in the diesel passenger fleet segment. For other countries, it is also well known that diesel cars are heavier than petrol cars and it appears that separate comparison of BEVs with diesel and petrol cars may lead to a more accurate picture of relevant vehicle categories.

It is noted that the difference in weight between ICEVs and BEVs is likely to change as battery energy density improves substantially over the coming years and light-weight materials are increasingly used by vehicle manufacturers. An increase in energy density would lead to either smaller and lighter batteries, or an increased drive range, or a combination of the two. These developments would likely lead to lower non-exhaust PM emissions. Another factor is the flexibility in BEV design that has not yet been fully utilised and can lead to further weight reduction. Whereas ICEV design needs to accommodate a cylinder engine block, a crankshaft and transmission, BEVs are not as constrained and can have multiple small motors placed in flexible configurations (e.g. wheel hubs), or diverse into a variety of electric (light-weight) road vehicles such as e-scooters and e-bikes.[35]

5.2 Regenerative braking

BEVs typically use a combination of regenerative braking and conventional friction braking, so called brake blending.[27] The amount of regenerative braking depends on personal driving style and road conditions, which are difficult to predict.[36]

The purpose of regenerative braking is to increase range by recuperating some of the energy normally lost when braking. An additional benefit is that brake wear emissions are reduced significantly, and estimates range widely from 25-95%.[2,28,9]

For instance, it has been assumed that BEVs require, on average, about two-thirds less braking than ICEVs due to regenerative braking.[9] The assumption is based on analysis of service times of brake pads on Teslas, BMW i3s, and Nissan Leafs. This demonstrated that, on average, the brake pads last roughly two-thirds longer than on ICEVs. Another research study compared the front wheels of a 1 year old conventional ICEV and a 1 year old BEV and noted that even after 32,000 km the wheel and brakes of the BEV still looked brand new, indicating very little brake wear had occurred.[28]

5.3 Driving behaviour does not change

Driving behaviour and traffic conditions are a major factor with respect to the actual levels of tyre, brake and road wear, and thus non-exhaust PM emissions.[31] For instance, an aggressive driving style will tend to result in more rapid and uneven tyre wear than a more restrained driving style. Driving behaviour, in particular the frequency and severity of braking events, is also an important determinant of brake wear.[28]

Most of the reviewed research studies have assumed that driving behaviour is constant and does not change when comparing ICEVs with BEVs. This is unlikely to be the case. It has been observed that driving behaviour significantly changes when people swap an ICEV for a BEV. For instance, in one study route choice behaviour changed substantially, suggesting that BEV users change their usual routes to try to balance the trade-off between travel time and energy consumption.[37] BEV drivers are also reported to have better eco-driving and 'energy saving' behaviour (e.g. less braking, smoother driving), which would also reduce non-exhaust PM emission rates.[2,30 35] Whether this translates partially or completely to future increased uptake of BEVs is unclear at this stage.

5.4 Uncertainty in resuspended road dust

Road dust comes from a wide variety of sources, indeed virtually any man made (exhaust, non-exhaust) or natural source (salt, grit, biogenic, geogenic) can deposit dust particles on the road.[11,14] Research studies have found that road dust is largely made up of soil-derived minerals (reflecting local geology) and non-exhaust PM, with only a small fraction coming from plant matter and exhaust PM.[14,30]

Resuspended road dust can cause high and extremely variable PM emissions, and associated concentration levels, depending strongly on dust loading on the surface.[38] For instance, non-exhaust PM emissions on unpaved roads are dominated by resuspension of road dust and orders of magnitude higher then emissions on paved roads.[39] A research study reported that the measured average real-world PM emission factor for paved roads was 26 mg/km and 24,000 mg/km for unpaved roads.[22]

Resuspension is particularly difficult to quantify as it is a function of several factors, including season, precipitation, road moisture content, the type of road and road condition. For instance, porous pavements seem to retain deposited dust better than dense pavements, thus leading to lower emissions due to resuspension compared to pavements with a dense structure (e.g. asphalt, concrete).[38]

In addition, there is the issue of double counting. Due to their similar chemical composition and high correlation in time, emissions from road wear and resuspended road dust are not easy to distinguish in field data.[12] PM due to resuspension processes may already have been attributed to other emission sources such as road wear.[11] As a consequence, the contribution of resuspended road dust to emissions and ambient concentrations is location specific and varies widely from about 10% to 90%.[20,40,41]

From a vehicle perspective, resuspension of road dust is mainly a function of vehicle-induced turbulence, which in turn depends on vehicle size, vehicle weight, aerodynamic properties and vehicle speed. Indeed, heavy-duty vehicles (trucks, buses) are responsible for most of resuspended road dust in real-world conditions.[42]

Although it has been assumed that BEVs increase resuspended road dust emissions because they are assumed to be heavier [30], it appears that other contributing factors should be considered more carefully. For instance, without an exhaust system, the undercarriage of BEVs is generally flatter compared to ICEVs, which should improve aerodynamics and reduce resuspended road dust emissions.

Quantifying the relative impact of BEVs on resuspended road dust is highly speculative at best in the absence of empirical data and could either be reduced or increased. Indeed, the large uncertainty has led many studies to omit this aspect altogether, effectively assuming BEV and ICEV have an equivalent impact on resuspended road dust. Given that diesel cars are generally heavier than BEVs in Australia, it seems likely that diesel cars and SUVs will create more resuspended dust than BEVs.

5.5 Exhaust impacts - atmospheric chemistry

The discussion of non-exhaust PM emissions and associated PM concentrations in ambient air is further complicated by dynamic and complex chemistry processes in the atmosphere. PM concentrations are not only the result of direct emissions of PM_{10} and $PM_{2.5}$ (primary aerosols), but also the result of particles formed in the atmosphere from gaseous precursors emitted by fossil-fuelled vehicles, the so-called secondary aerosols (PM).

Secondary PM may significantly contribute to total PM concentration levels with reported values between 25% and 50% of ambient PM concentration levels.[41,43,44] Secondary PM is formed in the atmosphere when gases such as (semi) volatile organic compounds (VOCs), nitrogen oxides (NO_x), sulphur dioxide (SO_2), and ammonia (NH_3) undergo a series of physical and chemical reactions under the influence of sunlight. In terms of fossil-fuelled road traffic, NO_x appears to be the most important precursor of secondary PM in urban areas, although the impact of fossil-fuelled vehicles on secondary organic PM may have been substantially underestimated.[44,45]

Measurement and source allocation of secondary PM is very challenging to quantify accurately. The atmospheric chemical reactions that form secondary PM are highly nonlinear and time dependent.[43,46] As a consequence, the impact of BEVs on secondary aerosol formation is generally ignored.

However, it could be important. BEVs do not emit gaseous precursors as they have no exhausts or fuel systems. This means BEVs will potentially significantly reduce the creation of secondary aerosols, and lower urban PM concentration levels. But if a substantial amount of electricity is generated by fossil-fuel fired power stations, as is currently the case in Australia, secondary aerosols (e.g. precursor SO_2 for sulphates) can be formed from indirect pollutant emissions from coal-fired power stations.

International studies that have assessed the impacts of large power stations on urban air quality have reported small contributions to city $PM_{2.5}$ concentrations (both primary and secondary) in the order of 5%.[47,48] But the impact largely depends on local factors such as location of the power station (including distance to city), local geography and meteorology.

In Australia, an assessment of the modelled impact of natural and anthropogenic sources on air quality in the Greater Metropolitan Region of Sydney concluded that motor vehicles and power stations contributed 17% and 11%, respectively to local $PM_{2.5}$ exposure (primary and secondary PM combined).[49] Another study for the same region concluded that motor vehicles are only slightly more important source (8%) than power stations (7%) regarding population weighted average $PM_{2.5}$ concentrations.[50] The results may differ for other metropolitan regions in Australia, since the impact of power stations on urban air quality depends on the power station emissions (fuel, emission control) and distance to urban areas, among other things.

Nevertheless, the studies suggest that even though Australian BEVs are largely powered by electricity generated by coal-fired power stations, electrification of the on-road fleet will likely improve urban air quality with respect to PM exhaust emissions (fossil-fuelled tailpipe versus power station stack) and secondary PM, as compared with the current fossil-fuelled on-road fleet.

Of course, BEVs have the potential for deep reductions in emissions and thus large improvements in local air quality as Australia progressively moves to sustainable electricity generation (solar, wind, thermal, etc.). Fossil-fuelled vehicles do not have this potential.

6. Quantifying the impact of BEVs on non-exhaust emissions

When reviewing the available data on non-exhaust PM emissions, the scarcity of empirical data became clear. Firstly, there are only a limited number of studies from which quantitative information could be extracted. This is perhaps surprising given the contribution of non-exhaust PM emissions to total PM emissions from motor vehicles. Secondly, there is no standardised test protocol for measurement of non-exhaust PM emissions (AQEP, 2019). A range of methods have been used, including direct measurement through enclosure of brakes [e.g. 19,51], on-road sampling [e.g. 12,22,26,41,52], laboratory/on-road road simulators [e.g. 25,38,55], indirect measurement through road-side air quality monitoring [e.g. 39,40,42,53], and tunnel studies [e.g. 15,24]. Quantitative data for specifically for BEVs is largely absent.

With only limited quantitative empirical data, a defensible statement regarding the impact of BEVs on non-exhaust PM emissions requires inclusion of a measure of uncertainty. To do this a database of published emission factors (mg PM/km) was created from the available scientific literature. The rationale being that the range of published average emission factors, across a range of studies, reflect, to some extent, the real-world variability in non-exhaust PM emissions.

Figure 2 shows the resulting emission factor distributions for tyre wear, brake wear and road wear. The large variability is visible in the data.

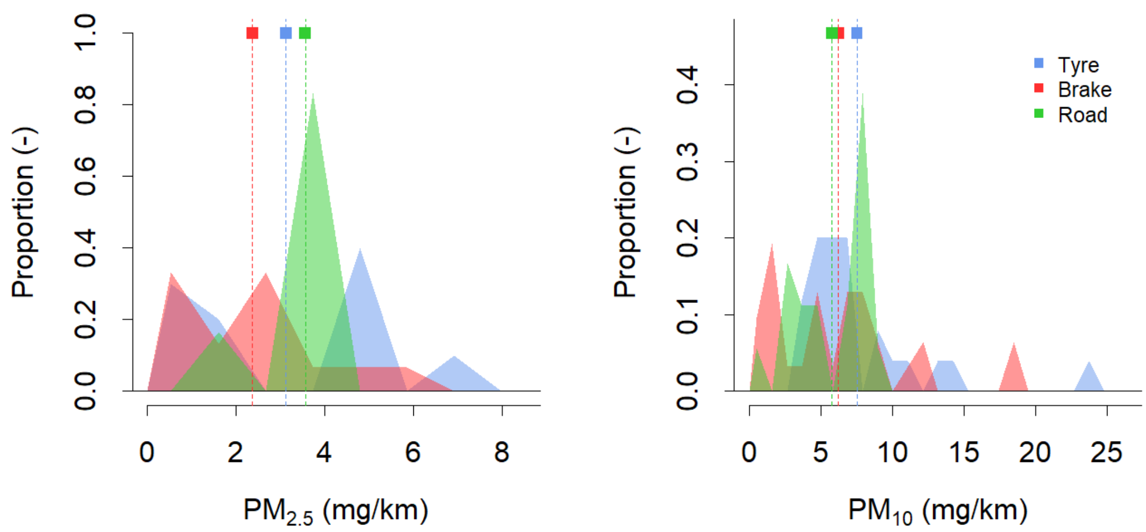

Figure 2 – Mean non-exhaust PM emission factor distributions (paved roads) from international literature, bootstrap grand mean values are shown as squares at the top of the charts.

It is noted that the emission factor distribution for resuspended road dust is not included in Figure 2, as it would make the figures hard to interpret due to the wide range of emission factor values. As was discussed in section 5.4, resuspended road dust emission rates are extremely variable. For instance, the PM_{10} emission factors for resuspended road dust on paved roads, published in international literature, varied from 0.3 mg/km to 780 mg/km, varying with more than 3 orders of magnitude. In comparison, emission factors for the other wear aspects (tyre, brake, road) still vary quite substantially, but to a lesser extent, with a variability of 1 to 2 orders of magnitude.

It is clear from Figure 2 that the mean emission factor distributions are not symmetric. In addition sample sizes are small (varying from 6 to 31). Therefore conventional computation of the confidence interval of the (grand) mean [5] is not appropriate. A bootstrap simulation [6] was conducted in R to estimate the grand mean and associated non-symmetric 95% confidence intervals (CI) for each non-exhaust PM aspect.[54] The results are shown in Table 1.

Table 1 – Mean Non-exhaust PM emission factors for ICEV PVs (Bootstrap 95% CI, n = sample size).

Non-Exhaust PM Aspect	$PM_{2.5}$ (mg/km)	PM_{10} (mg/km)
Tyre Wear	3.1 (1.8 – 4.5, n = 10)	7.6 (6.1 – 9.3, n = 25)
Brake Wear	2.4 (1.6 – 3.1, n = 15)	6.2 (4.7 – 7.9, n = 31)
Road Wear	3.6 (2.6 – 4.1, n = 6)	5.8 (4.7 – 6.8, n = 18)
Resuspension	7.2 (3.8 – 11.4, n = 13)	94.4 (37.5 – 175.6, n = 22)

It has been reported that non-exhaust PM emissions, not including resuspended road dust, are approximately equally split between brake wear, road wear and tyre wear [30,36], but that the relative contributions can vary substantially depending on local situations [56,57]. Table 1 shows similar results. Note that Table 1 shows the 'mean of means' (and associated confidence intervals), which can be interpreted as a typical value for each aspect of wear related PM emissions and resuspension.

As a next step, the emission distributions shown in Figure 2 were used in a probabilistic analysis to estimate the emission factor distribution of non-road PM exhaust emissions for (fossil-fuelled) ICEV vehicles ($e_{NE,ICEV}$) and BEVs ($e_{NE,BEV}$).

The non-exhaust PM emission factors are computed with two simple linear models:

$$e_{NE,ICEV} = e_{tyre,ICEV} + e_{brake,ICEV} + e_{road,ICEV} \; (+ e_{resusp,ICEV})$$

$$e_{NE,BEV} = \gamma_{tyre}\, e_{tyre,ICEV} + \gamma_{brake}\, e_{brake,ICEV} + \gamma_{road}\, e_{road,ICEV} \; (+ \gamma_{resusp}\, e_{resusp,ICEV})$$

Using these models, $e_{i,j}$ can be used to represent an emission factor (mg/km) for non-exhaust process *i* (brake, tyre, road, resuspension) and vehicle type *j*. The model inputs $e_{i,j}$ are specified as empirical emission factor distributions (Figure 2). Similarly, γ_i can be used to represent a BEV correction factor distribution for non-exhaust process *i*. Given the lack of empirical data a uniform distribution (equal probability between end points) was used using plausible minimum and maximum values, sourced from the available literature. The uniform distribution for γ_{tyre} was defined with 0.75 and 1.25 (section 5.1). The uniform distribution for γ_{brake} was defined with 0.25 and 0.95 (section 5.2). Resuspension of

[5] mean ± coverage factor (t-value) × standard error (standard deviation / √ sample size).

[6] Random resampling of the distributions with replacement (n = 10,000).

road dust was excluded from the calculation given the extreme uncertainty associated with this aspect of non-exhaust PM emissions, as well as the risk of double counting.

A Monte Carlo simulation was used to propagate the uncertainty reflected in the input distributions to the model outputs $e_{NE,ICEV}$ and $e_{NE,BEV}$.[58] This way not only expected values are estimated, but also the associated uncertainty. The process is a mathematical analogue of an experiment which is repeated many times to provide an accurate estimate of the result.

Random samples were taken one-at-the-time from the input distributions, and repeated many times (100,000), to create a probability distribution of $e_{NE,ICEV}$ and $e_{NE,BEV}$. This way, the likely range of non-exhaust PM emission factors is estimated for both fossil-fuelled ICEVs and BEVs.

The results are shown in Table 3 and Figure 3 and 4 (box-plots, next page).

Table 3 – Mean Non-exhaust PM emission factors for ICEV and BEVs (Monte Carlo simulation, 95% CI).

Technology	$PM_{2.5}$ (mg/km)	PM_{10} (mg/km)
ICEV	9.0 (3.4 – 14.1)	19.6 (9.4 – 35.9)
BEV	8.0 (2.9 – 13.2)	17.1 (8.6 – 32.7)
Difference	-0.9 (-0.1 – -3.1)	-2.5 (-0.1 – -9.0)

Table 3 shows that, accounting for variability in reported mean emission factors for brake wear, tyre wear and road wear, BEVs are expected to reduce non-exhaust $PM_{2.5}$ and PM_{10} average emission factors for passenger vehicles with 95% confidence.

The weight of evidence suggests that BEVs:

- reduce $PM_{2.5}$ non-exhaust emission rates with 1% to 34% (11% on average), and
- reduce PM_{10} non-exhaust emission rates with 1% to 46% (13% on average).

It is noted that the simulation only considered the 'between study' variability/uncertainty, and not the 'within study' variability/uncertainty, as this information was not available. If so, a meta-analysis technique (random effects analysis) could have been applied.

Therefore, the uncertainty will be larger than modelled above, and this will be compounded since a number of factors could not be considered (resuspended dust, driving behaviour change, atmospheric chemistry) for both practical reasons and lack of data, as was discussed previously.

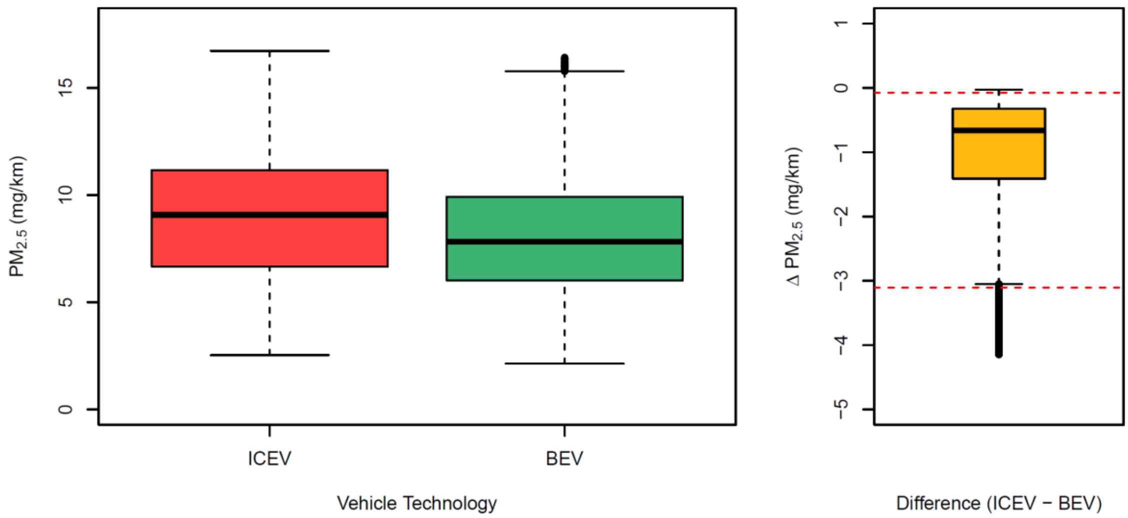

Figure 3 – Box-plots showing Monte Carlo simulation results for non-exhaust $PM_{2.5}$ emission rate distributions by technology class (left) and difference between class (right, yellow).

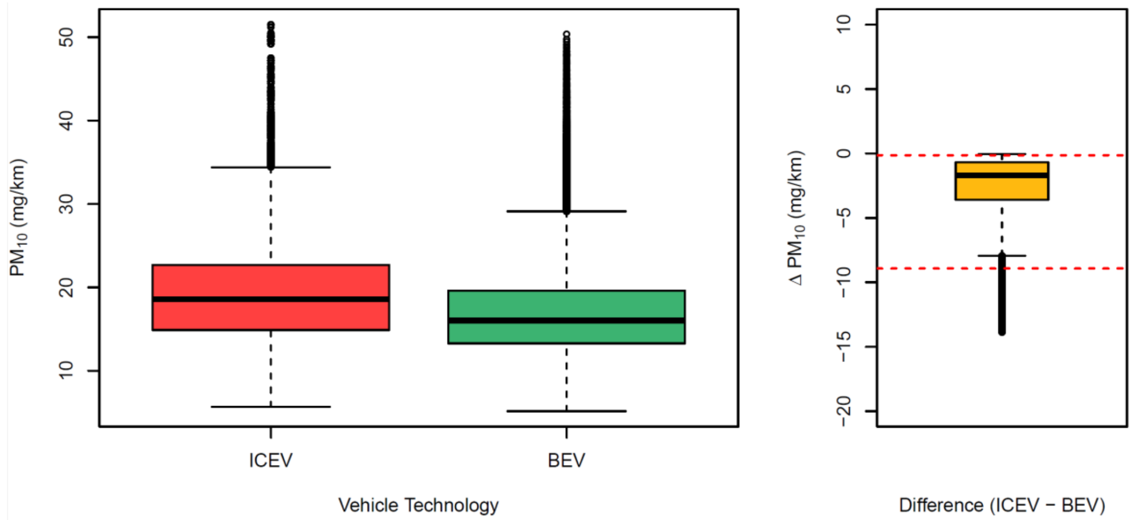

Figure 4 – Box-plots showing Monte Carlo simulation results for non-exhaust PM_{10} emission rate distributions by technology class (left) and difference between class (right, yellow).

7. Conclusions and Concluding Remarks

BEVs have several clear benefits over ICEVs in terms of greenhouse gas emissions and local air pollutant exhaust emissions.[3] The only local air pollutants that all on-road vehicles produce, including BEVs, are non-exhaust PM emissions due to brake wear, tyre wear, road wear and resuspended road dust. The extent to which BEVs will have an impact on non-exhaust PM emissions contributions is unclear and subject to debate.

Non-exhaust PM emissions are important. They generally dominate total PM (mass) emissions from road traffic. This study estimated a contribution of 70-80% of non-exhaust PM tot total (mass) PM nationally for the Australian on-road fleet in 2020. At a local scale there is, however, a lot of variability in these contributions. Nevertheless, even at a local scale studies have shown that non-exhaust PM emissions are generally at least as important as vehicle exhaust PM emissions. Further research into particle number PN emissions (exhaust and non-exhaust) is recommended.[e.g. 52]

This study has looked into the five main points of contention within scientific literature regarding non-exhaust PM emissions from BEVs.

1. *BEVs are heavier than ICEVs, and therefore generate more non-exhaust PM emissions*. There is consensus that heavier vehicles create more non-exhaust PM emissions. However, this study has identified debatable methods in studies that have arrived at the conclusion regarding BEVs being heavier than ICEVs. It also found that, at least in Australia with a relatively large and heavy passenger fleet, BEVs and petrol ICEVs have similar weight. In fact diesel ICEVs (passenger vehicles) are significantly heavier (25-30%). The discussion of the impact of vehicle weight on non-exhaust PM is therefore incorrectly focussed on BEVs, and should instead be focussed on diesel cars and SUVs. It is like watching the ball and missing the gorilla.

2. *BEVs use regenerative braking, which reduced non-exhaust PM emissions*. This is correct, however the use of regenerative braking in the real-world is uncertain and requires further study. As a consequence, the positive impact of regenerative braking on non-exhaust PM emissions is also uncertain, as is the reduction in brake wear emissions, which ranges from 25% to 95%.

3. *BEV driving behaviour is the same as ICEV driving behaviour*. Driving behaviour is a major contributing factor to the actual level of non-exhaust PM emissions. The reviewed research studies incorrectly assume that driving behaviour does not change when people swap an ICEV for a BEV. BEV drivers may change route choice and will likely exhibit a smoother (energy-saving driving) driving style, which will reduce non-exhaust PM emissions.

4. *Resuspended road dust emissions are the same for BEVs and ICEVs*. No data or information was found to suggest otherwise, although this does not mean there are no differences. The main issue in assessing this is the extreme local variability in resuspended road dust emissions and the risk of double-counting of wear emissions. Quantifying the relative impact of BEVs on resuspended road dust is therefore highly speculative. Given that diesel cars are generally heavier than BEVs in Australia, it seems likely that diesel cars and SUVs will create more resuspended dust than BEVs.

5. *Atmospheric chemistry is not considered*. PM concentrations are not only the results of direct PM emissions from vehicle exhausts and wear (primary aerosols), but also the result of particles formed later in the atmosphere from air pollutant emissions from fossil-fuelled vehicles (secondary aerosols). Dynamic and complex chemistry processes in the atmosphere complicate the assessment of impacts of secondary PM, so often this aspect is not considered in research studies. A review of the literature suggests that even in countries where BEVs are largely powered by coal-fired power stations (e.g. Australia), electrification of the on-road fleet will improve urban air quality with respect to total ambient PM concentrations (primary and secondary combined), as compared with the current fossil-fuelled on-road fleet.

The main points above demonstrate the complexity in assessing the impact of BEVs on non-exhaust PM emissions, and show that the conclusions from research studies are often based on various (often debatable) assumptions leading to uncertainty and contention.

This study therefore conducted a probabilistic assessment of the expected impact of BEVs on non-exhaust PM emissions, taking ICEVs as the base case for comparison purposes. This approach explicitly considers and reflects the variability and uncertainty in non-exhaust PM emissions. Emission factor distributions were created by extracting mean emission factors from the available literature. What became clear is the scarcity in empirical (non-exhaust) PM emissions data and information. Quantitative data specifically for BEVs is even more scarce, largely absent in fact. It is therefore essential that the uncertainty in non-exhaust PM emission estimates is considered in any statements or analysis.

The TER analysis estimates that BEVs (passenger vehicles) *reduce* non-exhaust $PM_{2.5}$ and PM_{10} emissions with 95% confidence.

BEVs are expected to reduce fleet average $PM_{2.5}$ non-exhaust emission rates somewhere between 1% to 34% (on average 11%), and reduce PM_{10} non-exhaust emission rates somewhere between 1% to 46% (on average 13%).

A focus on the impacts of BEVs on non-exhaust PM emissions appears therefore unjustified.

In a time where a move to zero emission transport is urgently required, non-exhaust PM emissions should not be used as an (invalid) argument against rapid electrification of the on-road fleet. This study suggests that instead the focus should be on reducing the sale of (heavy) diesel passenger vehicles, which are expected to be a more important factor in non-exhaust PM emissions, at least in Australia.

Finally, initial efforts are made overseas to reduce non-exhaust PM emissions from BEVs. For instance, a German company has developed an external particle filter system for BEV to collect PM emitted from brakes, tyres and the road surface.[59] The idea is that this way, a BEV could become truly a real "zero emission" vehicle.

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