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Exhaust and non-exhaust emissions from conventional and electric vehicles: A comparison of monetary impact values

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Abstract: The switch to electric vehicles (EVs) has been incentivised by governments all over the world to reduce the use of fossil fuels and improve air quality. However, whether such a move could effectively lower the levels of pollutants as much as expected is still controversial. This study estimates the impact values of exhaust and non-exhaust emissions emitted from internal combustion engine vehicles (ICEVs) and their equivalent EVs from an economic-environmental perspective, expressed as monetary impact values, so as to ascertain the environmental effect of the switch to equivalent EVs from ICEVs. These monetary impact values were calculated according to the emission factors and damage costs of these pollutants. The results indicate that the particulate matter (PM) monetary impact values of equivalent EVs may exceed those of ICEVs, which depends primarily on the extent of regenerative braking and road type. The monetary impact values of total pollutants decrease for the move from diesel passenger cars to their equivalent EVs with 0% regenerative braking. For the conversion of petrol passenger cars to their equivalent EVs with 0% regenerative braking, however, the total monetary impact values increase on both urban and rural roads. These results can be useful for the economic-environmental assessment of vehicle exhaust and non-exhaust emissions.

Keywords: Monetary impact values; Exhaust emissions; Non-exhaust emissions; Electric vehicles; Conventional vehicles

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1. Introduction

Air quality has always been a topic of ongoing public policy concerns. Vehicle exhaust, such as nitrogen oxides (NO_x), exhaust particulate matter (PM), carbon monoxide (CO), volatile organic compounds (VOCs), and non-exhaust emissions, such as tyre wear, brake wear, road wear particle emissions and resuspension of road dust, are the main contributors to air pollutions and are recognised as critical risk factors in the air in terms of harm to human health (García-Contreras et al., 2020; Kousoulidou et al., 2008; Pant and Harrison, 2013; WHO, 2020). The switch to electric vehicles (EVs) from internal combustion engine vehicles (ICEVs referring to conventional vehicles) has been proposed and incentivised to improve air quality (Calef and Goble, 2007; Mock and Yang, 2014; Murrells and Pang, 2013). Compared to ICEVs, the advantage of EVs is that they generate no exhaust emissions in real driving. For instance, Singh and Strømman, (2013) found that the introduction of EVs to passenger vehicle fleets would reduce exhaust emissions of NO_x, SO_2 and PM. The assessment of lifecycle greenhouse gas emissions performed by Ahmadi, (2019) found that greenhouse gas emissions from EVs were approximately half of that from ICEVs. In addition, Ahmadi et al., (2018) optimised electric drive vehicle selection based on lifecycle emissions. The results showed that plug-in hybrid electric vehicles were the optimal solution compared to conventional gasoline vehicles, hybrid vehicles and full-battery EVs during a wide range of vehicle miles travelled when lifecycle greenhouse gas emission was the sole objective. (Shamsi et al., 2021) revealed that every kilogram of H_2 was used for fuel cell electric vehicles, avoiding 11.09 kg CO_2 emitting the atmosphere compared to equivalent conventional vehicles. In the case of PM emissions, however, Hooftman et al., (2018) revealed that EVs would generate more non-exhaust emissions due to the heavier weight relative to ICEVs and that the switch to

EVs might not lead to a significant reduction in the levels of PM. In the study by Soret et al., (2014), the effect of EVs on air quality was modelled, and it was discovered that fleet electrification could not considerably decrease PM emissions due to the generation of more non-exhaust emissions. Timmers and Achten (2016) revealed that the non-exhaust emissions from equivalent EVs, even with 100% regenerative braking, might exceed all PM emissions emitted from the conventional vehicles. It is also important to note that if the electricity needed to fuel the battery is not produced by photovoltaic and wind power plants, the pollutants emitted during the electricity production and the effect of various charging rates on emission have to be considered (Maroufmashat et al., 2020).

Conventional passenger cars emit emissions through exhaust and non-exhaust pathways, while EVs only generate non-exhaust emissions. As a result, the full estimation of the switch to equivalent EVs from conventional passenger cars needs to take into consideration the multiple criteria of different units, dimensions, and the importance of all pollutants. To make the estimation easily interpretable, unifying the impacts of pollutants into a single unit is necessary. Pizzol et al., (2015) pointed out that monetary impact value can convert the social, environmental and biophysical impacts into monetary units to identify the economic values of non-market goods, so as to realise the cross-comparison among various impacts. Thus, the monetary impact value is introduced as an indicator since the health, economic, or environmental effects of exhaust and non-exhaust emissions can be converted into monetary units, which can be directly compared.

Jaramillo and Muller, (2016) calculated the monetary values of $PM_{2.5}$, SO_2, NO_x, NH_3, and VOCs from electric power generation, coal mining, and oil refineries. Dong et al., (2019) estimated the monetary value of greenhouse gas emissions (GHG) from anthropogenic sources and found human health damages contributed to 70%−79% of the GHG monetary values. Yildiz et al., (2019) analysed the environmental pollution cost of a diesel engine fuelled with biodiesel and diesel fuels. The results showed that a diesel engine fuelled with biodiesel generated a higher environmental pollution cost than that with diesel. Kusiima and Powers (2010) monetised the life cycle environmental and health externalities related to ethanol production. The results showed that the mean external cost for producing 1-litre ethanol ranged from $0.07 for forest residue to $0.57 for ethanol production from corn, and the external costs of PM_{10}, NO_X, and $PM_{2.5}$ were the most significant contributors to these costs. To the best of our knowledge, monetising the exhaust and non-exhaust emissions from ICEVs and EVs is not available in the current literature. In addition, exhaust and non-exhaust emissions from passenger cars are significant contributors to air pollutions (Goel and Kumar, 2014), affecting human health, visibility and climate change (Peng et al., 2016; Ubando et al., 2021). Thus, the monetisation of these emissions is beneficial for the regulatory authorities and policymakers to design the mitigation strategies and to compute their contributions and impacts on public health and local air quality.

In this context, the purpose of this study is to evaluate the total impacts of exhaust and non-exhaust emissions emitted to the environment, expressed as monetary impact values. Monetary impact value assessment of exhaust and non-exhaust emissions emitted from ICEVs and EVs is based on the emission factors (EFs) and damage values of the corresponding emissions. Furthermore, the total monetary impact values of emissions produced from the ICEVs and their equivalent EVs were compared to ascertain the environmental effect of the switch to EVs. In the current work, only conventional passenger cars and their equivalent EVs are examined because the statistics of these vehicle weights are available in order to obtain emission factors.

2. Methods

2.1. Mass estimation of ICEVs and EVs

Non-exhaust airborne particles from traffic are generated from tyre wear, brake wear, and road surface wear, and the resuspension of deposited material that already existed on-road owing to vehicle-induced turbulence (Amato et al., 2011). Each of the non-exhaust airborne particles' sources is closely related to vehicle weight (Simons, 2016; Timmers and Achten 2016; Timmers and Achten, 2018; Wang et al., 2017). The conversion of ICE passenger car to an equivalent EV causes an increase in vehicle weight due to the heavier battery pack compared to an equivalent conventional engine. To evaluate the additional non-exhaust emissions of the transition from the ICEVs to equivalent EVs, ICEV-EV pairs were chosen from an internet database (Chapple, 2017). For each pair of selected either petrol or diesel passenger car and equivalent EV, vehicle parameters were matched as closely as possible, and their output powers were within 13% of each other. 21 pairs for petrol cars and equivalent EVs as well as 15 pairs for diesel cars and equivalent EVs, were chosen and listed in Tables S1 and S2 of the supplemental materials. The mean weight calculated for ICE petrol cars is 1412 kg, and their equivalent EVs are 323 kg heavier, corresponding to a 23% increase in weight. Likewise, ICE diesel cars have a mean weight of 1488 kg, and their equivalent EVs are 263 kg heavier with an 18% increase in weight. Faria et al., (2012) calculated that, on average, equivalent EVs are 20% heavier than their ICEVs. In the calculation of Timmers and Achten (2016), it was found that the mean weight increase was 280 kg for the switch to equivalent EVs from ICE passenger cars, corresponding to a 24% increase in vehicle weight. Beddows and Harrison, (2021) reported that the vehicle average mass from ICE passenger cars converting to equivalent EVs increased by 300 kg (21%). These data in the literature are in accordance with the current results.

2.2. Calculation of emission factors

The emission factor (EF) values were calculated using the mathematical model proposed by Beddows and Harrison, (2021). This model building includes the following steps: (1) the non-exhaust $PM_{2.5}$ and PM_{10} EFs for various vehicle types on various types of roads employed in national inventories were adopted (AQEG, 2019); (2) the various types of vehicle mass was evaluated; (3) a non-linear least-squares fit of the data was done to determine the separate correlations between EFs and vehicle mass for each tyre wear, brake wear, road wear and resuspension of road dust using equation (1); (4) the masses of ICE petrol and diesel passenger cars and the equivalent EVs were estimated; (5) the EFs for ICE passenger cars and the equivalent EVs on urban, rural, motorway roads were obtained. The present study was done under the assumption that the obtained results represent the average values of exhaust and non-exhaust emissions from ICEVs and EVs within a fleet, including different driving conditions.

$$EF = b \cdot W_{ref}^{\frac{1}{c}} \tag{1}$$

where W_{ref} equals to vehicle mass divided by 1000 kg, and the b (mg km^{-1} veh^{-1}) and c (no unit) are regression coefficients listed in Table S3 of the supplemental materials.

2.3. Calculation of monetary impact values

To obtain the monetary impact values of exhaust and non-exhaust emissions, the damage cost values of these emissions and corresponding emissions factors are required. Table 1 summarises the mean damage cost values of vehicle exhaust and non-exhaust emissions, alongside the low and high sensitivities, according to recommendations from the latest UK guidance on damage cost values in air quality appraisal (GOV, 2020). The damage costs were calculated in the latest UK guidance using the impact pathways approach (IPA). The IPA includes five stages: (1) Make sure the policy intervention; (2) Model the dispersion of air pollutant emissions; (3) Understand variations in environmental pollutant concentrations using dispersion modelling; (4) Calculate how these variations in concentrations influence different impact pathways related to health, economy and environment by means of concentration-response functions; (5) Value these damage costs using a single monetary metric (GOV, 2020). It can be seen in Table 1 that there is the highest damage cost for $PM_{2.5}$, followed by PM_{10}, NO_x, ammonia (NH_3) and VOC. The non-regulated pollutant, NH_3, was taken into account because the damage cost value and emission factor of NH_3 from the ICEVs were high (Brown et al., 2018). It is worth mentioning that CO is not involved in the estimation primarily due to the small damage cost (Wang and Santini, 1995).

Table 1. Mean damage cost values for pollutants emitted, alongside the low and high sensitivities.

Pollutants emitted	Mean damage cost (£/ton)	Damage cost sensitivity range (£/ton): low	Damage cost sensitivity range (£/ton): high
$PM_{2.5}$ from road transport	81,518	17,567	252,695
PM_{10} from road transport	54,862	11,823	170,063
NO_x from road transport	9,066	817	34,742
NH_3	7,923	1,521	37,611
VOC	102	55	205

Compared to ICEVs, EVs can reduce brake wear emissions effectively due to the possibility of installing a regenerative braking system in the vehicle. Various abatement rates of brake wear emissions have been reported due to regenerative braking, as shown in Fig. 1. For instance, Ligterink et al., (2014) suggested that regenerative braking could reduce brake wear emissions up to 95%. Althaus and Gauch (2010) revealed a 90% reduction in brake wear emissions through regenerative braking in the light of driving behaviour analyses. Del Duce et al., (2016) pointed out that brake wear emissions would be reduced by 80% due to regenerative braking. In a study by Hooftman et al., (2016), it was estimated that regenerative braking could decrease brake emissions by up to 66%. Van Zeebroek and De Ceuster (2013) indicated a 50% reduction in brake wear emissions due to regenerative braking. A more conservative prediction was given by Nopmongcol et al., (2017), who pointed out that regenerative braking led to a 25% reduction in brake wear emissions. Based on the average values of regenerative braking in the literature above, the EVs with 68% regenerative braking are used in this paper.

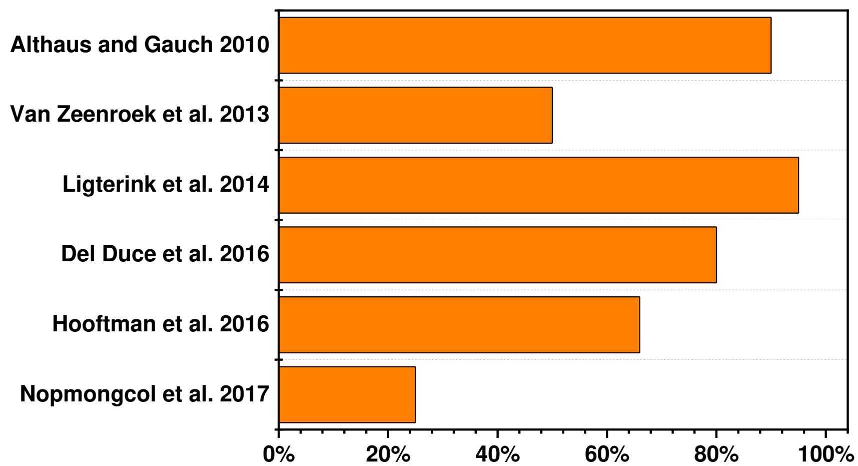


Fig. 1. The percentage reduction in brake wear emissions due to regenerative braking, based on the previous research work (Althaus and Gauch, 2010; Del Duce et al., 2016; Hooftman et al., 2016; Ligterink, NE et al., 2014; Nopmongcol et al., 2017; Van Zeebroek and De Ceuster, 2013)

The monetary impact values (IV) for ICEVs and EVs are evaluated based on the damage costs (DC) (see Table 1) of exhaust and non-exhaust emissions and their emission factors (EF). The difference in the monetary impact values of emissions from ICEVs and EVs is the effect of exhaust emissions and regenerative braking on monetary impact values. The total ICEV impact values were calculated by equations (2a-2d) (Beddows and Harrison, 2021), where the $EF_{ICEV_i}^{Non-exhaust}$ is the emission factor of *i* particle size class ($PM_{2.5}$ or PM_{10}) from non-exhaust emissions of ICEV. The total EV impact values ($IV_{EV}^{j\%\ RB}$) with *j*% regenerative braking (RB) are calculated by equations (3a) and (3d), where the $EF_{EV_i}^{j\%\,RB}$ is the emission factor of *i* particle size class ($PM_{2.5}$ or PM_{10}) from non-exhaust emissions of EVs with *j*% regenerative braking. The total impact values of emissions from EVs with 0% and 68% regenerative braking were evaluated on the assumption that the EVs with 0% and 68% regenerative braking would reduce 0% and 68% brake wear emissions (Beddows and Harrison, 2021).

$$IV_{ICEV} = IV_{ICEV}^{Exhaust} + IV_{ICEV}^{Non-exhaust} \tag{2a}$$

$$IV_{ICEV}^{Exhaust} = DC_{NO_x} \times EF_{NO_x} + DC_{VOC} \times EF_{VOC} + DC_{NH_3} \times EF_{NH_3} + DC_{PM_{2.5}} \times EF_{PM_{2.5}}^{Exhaust} \tag{2b}$$

$$IV_{ICEV}^{Non-exhaust} = DC_{PM_{2.5}} \times EF_{ICEV_{PM_{2.5}}}^{Non-exhaust} + DC_{PM_{10}} \times EF_{ICEV_{PM_{10}}}^{Non-exhaust} \tag{2c}$$

$$EF_{ICEV_i}^{Non-exhaust} = EF_{ICEV_i}^{TYRE} + EF_{ICEV_i}^{ROAD} + EF_{ICEV_i}^{RESUS} + EF_{ICEV_i}^{BRAKE} \tag{2d}$$

$$IV_{EV}^{j\%\,RB} = DC_{PM_{2.5}} \times EF_{EV_{PM_{2.5}}}^{j\%\,RB} + DC_{PM_{10}} \times EF_{EV_{PM_{10}}}^{j\%\,RB} \tag{3a}$$

$$EF_{EV_i}^{j\%\,RB} = EF_{EV_i}^{TYRE} + EF_{EV_i}^{ROAD} + EF_{EV_i}^{RESUS} + (1 - j\%) \times EF_{EV_i}^{BRAKE} \tag{3b}$$

3. Results and discussion

3.1. Emission factors of ICEVs and EVs

3.1.1. Non-exhaust emission factors

Fig. 2 shows the non-exhaust emission factors calculated for conventional petrol and diesel passenger cars and equivalent EVs. As expected, PM_{10} and $PM_{2.5}$ emission factors from tyre and brake wear of conventional passenger cars and their equivalent EVs on urban roads are higher than those on rural and motorway roads. Such a behaviour is closely associated with the higher frequency of acceleration and deceleration manoeuvres on urban roads compared to rural and motorway roads, causing an increase in tyre and brake wear emissions (Kwak et al., 2013; Yang et al., 2018). Wahid, (2018) revealed that the brake wear emissions was more prominent in urban cities due to the high braking frequency, especially during peak hours. In a review by Grigoratos and Martini (2015), it could be referred that brake wear emissions were severe in urban environments, contributing up to 55% by mass to total non-exhaust traffic-related PM_{10} emissions. In a simulation conducted by Cho et al., (2011), it was found that the acceleration and deceleration manoeuvres would cause a sharp increase in tyre wear emissions. Thus, there is a high frequency of acceleration and deceleration manoeuvres on urban roads than on rural and motorway roads, which leads to increased tyre wear emissions.

In Fig. 2, it can be observed that compared ICEVs, EVs emit more non-exhaust $PM_{2.5}$ and PM_{10} emissions. Such an increase in non-exhaust emissions is associated with the equivalent EVs possessing heavier weight relative to IECVs. The tyre wear and road abrasion are closely related to the normal force against the road and the friction coefficient between them (Rajamani et al., 2010). The normal force is proportional to vehicle weight, indicating that the increased EV weight would emit more tyre and road wear emissions. The brake wear emissions are generated by the friction between the brake pad and wheels and are highly dependent on friction energy (Woo et al., 2021). The friction energy is proportional to the vehicle weight and speed (Kakad et al., 2017; Timmers and Achten 2016). As a result, more friction energy is required to reduce vehicle speed as vehicle weight increases, generating more brake wear emissions. Vehicle-induced turbulence leads to the resuspension of road dust, which is closely related to vehicle weight and size (Timmers and Achten 2016). Thus, heavier EVs cause strong turbulence, leading to an increase in resuspension of road dust.

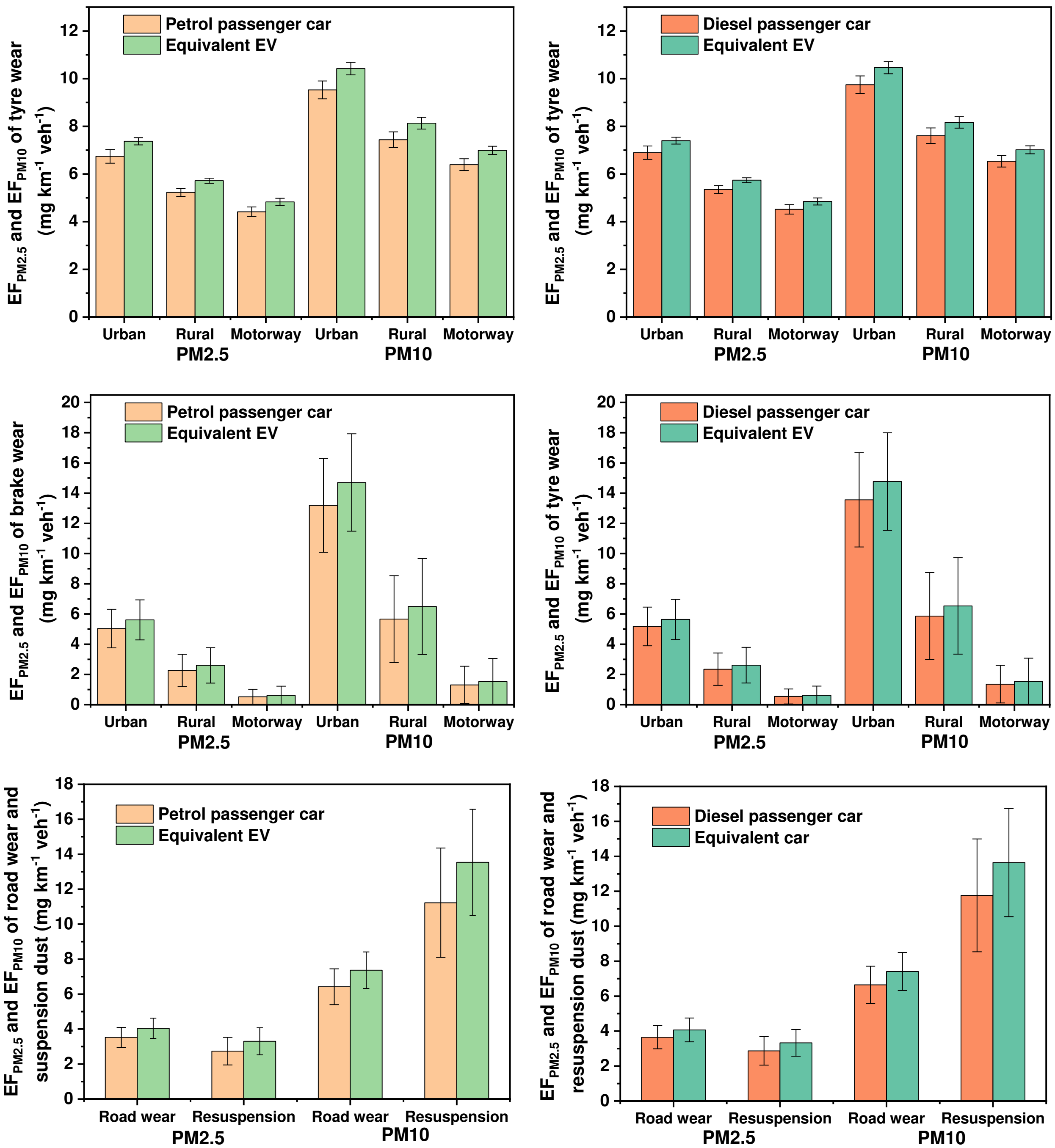


Fig. 2. Non-exhaust emission factors for petrol and passenger cars and their equivalent EVs.

Average PM_{10} non-exhaust emission factors on urban, rural, and motorway roads from tyre wear, brake wear, road wear, and resuspension dust for petrol passenger cars are 7.79, 6.72, 6.42, and 11.22 mg km^{-1} veh^{-1}, respectively. On average, $PM_{2.5}$ non-exhaust emission factors for petrol cars are 5.46, 2.61, 3.52, and 2.74 mg km^{-1} veh^{-1} generated from tyre wear, brake wear, road surface wear, and resuspension of road dust, respectively. Compared to petrol passenger cars, diesel passenger cars generate slightly higher non-exhaust emissions, which is mainly because of the heavier weight of diesel cars relative to petrol cars, as manifested in Tables S1 and S2 of the supplemental materials. Table 2 summarises the results of the PM_{10} and $PM_{2.5}$ EFs (mg km^{-1} veh^{-1}) from the tyre, brake, road wear and resuspension of road dust published in the literature and current study. Panko et al., (2013) reported that tyre PM_{10} EFs were 7.0 mg km^{-1} veh^{-1} through roadside study. Beddows and Harrison (2021) evaluated the PM_{10} and $PM_{2.5}$ EFs of tyre wear using a receptor modelling method. They found and that the mean values of PM_{10} and $PM_{2.5}$ EFs from passenger cars on urban, rural and motorway roads were 7.1 mg km^{-1} veh^{-1} and 5.0 mg km^{-1} veh^{-1}, respectively. The updated emission inventory by U.S. EPA calculated tyre PM_{10} EF value of 6.1 mg km^{-1} veh^{-1} (U.S. EPA, 2014). In the case of brake wear emissions, Garg et al., (2000) evaluated the brake wear emissions using a brake dynamometer study. They reported that brake PM_{10} and $PM_{2.5}$ EFs were 5.2 mg km^{-1} veh^{-1} and 2.3 mg km^{-1} veh^{-1}, respectively. In the study by Iijima et al., (2008), they stated brake PM_{10} EF value of 8.1 mg km^{-1} veh^{-1}. Recently, Piscitello et al., (2021) reported that the mean values of brake wear PM_{10} and $PM_{2.5}$ EFs were 7.4 mg km^{-1} veh^{-1} and 2.3 mg km^{-1} veh^{-1}, respectively. Timmers and Achten (2016) revealed road wear PM_{10} and $PM_{2.5}$ EFs of 7.5 mg km^{-1} veh^{-1} and 3.1 mg km^{-1} veh^{-1}, respectively. Beddows and Harrison (2021) obtained the mean PM_{10} and $PM_{2.5}$ EFs of road wear of 6.1 mg km^{-1} veh^{-1} and 3.3 mg km^{-1} veh^{-1} as well as resuspension PM_{10} and $PM_{2.5}$ EFs of 11 mg km^{-1} veh^{-1} and 2.7 mg km^{-1} veh^{-1}. Zhang et al., (2020) acquired the mean resuspension PM_{10} and $PM_{2.5}$ EFs of 20.8 mg km^{-1} veh^{-1} and 2.7 mg km^{-1} veh^{-1} using the chemical mass balance method. As discussed above mentioned, most of the non-exhaust EFs published in the literature are in agreement with the current results.

Table 2. Summary of the PM_{10} and $PM_{2.5}$ EFs (mg km^{-1} veh^{-1}) from tyre, brake, road wear and resuspension of road dust.

Non-exhaust emissions	PM_{10} EF	$PM_{2.5}$ EF	Data sources	City (Country)	Reference
Tyre wear	7.79[a] 7.97[b]	5.46[a] 5.59[b]	Receptor modelling	UK	Present work
	7.0		Roadside study	Pittsburgh (USA)	(Panko et al., 2013)
	7.1	5.0	Receptor Modelling	Birmingham (UK)	(Beddows and Harrison, 2021)
	6.1	2.9	Receptor Modelling	Breda (Netherlands)	(Timmers and Achten 2016)
	6.1		Emission inventory	USA	(U.S. EPA, 2014)
Brake wear	6.72[a] 6.93[b]	2.61[a] 2.69[b]	Receptor modelling	UK	Present work
	5.2	2.3	Brake dynamometer study	Michigan (USA)	(Garg et al., 2000)
	5.8		Brake dynamometer study	Maebashi (Japan)	(Iijima et al., 2008)
		0−5	Roadside study	Reno (USA)	(Abu-Allaban et al., 2003)
	6.20	2.47	Receptor modelling	Birmingham (UK)	(Beddows and Harrison, 2021)
	7.4	2.3	Receptor modelling	Torino (Italy)	(Piscitello et al., 2021)
Road wear	6.42[a] 6.65[b]	3.52[a] 3.65[b]	Receptor modelling	UK	Present work
		2−25	Roadside study	Reno (USA)	(Abu-Allaban et al., 2003)
	7.5	3.1	Receptor modelling	Breda (Netherlands)	(Timmers and Achten 2016)
	6.1	3.3	Receptor modelling	Birmingham (UK)	(Beddows and Harrison, 2021)
	7.75	4.05	Receptor modelling	Torino (Italy)	(Piscitello et al., 2021)
Resuspension of road dust	11.22[a] 11.77[b]	2.74[a] 2.87[b]	Receptor modelling	UK	Present work
	5.4−9.0	~	Roadside study	Barcelona (Spain)	(Amato et al., 2016)
	11	2.7	Receptor modelling	Birmingham (UK)	(Beddows and Harrison, 2021)
	20.8 ± 2.5	2.7 ± 0.9	Chemical mass balance	Tianjin (China)	(Zhang et al., 2020)
	~	3.6 ± 1.7	Chemical mass balance	Qingdao (China)	(Zhang et al., 2020)

Note: [a] Mean EFs for tyre, brake, road wear, resuspension from ICE petrol cars on urban, rural and motorway roads.
[b] Mean EFs for tyre, brake, road wear, resuspension from ICE diesel cars on urban, rural and motorway roads.

To further evaluate the effect of the transition from ICE petrol and diesel passenger cars to equivalent EVs on non-exhaust emissions, the increment and increased percentage in non-exhaust emission factors were calculated, and the calculated results are summarised in Table 3. The percentage increases in non-exhaust emission factors on urban, rural, and motorway roads are in the range of 9.73–20.59% for the switch to equivalent EVs from the petrol passenger cars and 7.33–15.94% for diesel passenger cars switching to the corresponding EVs. Compared to petrol passenger cars switching to the corresponding EVs, there are smaller percentage increases in non-exhaust emission factors of diesel vehicles. This is primarily because diesel vehicles have smaller weight increases relative to petrol vehicles, as illustrated in Tables S1 and S2 of the supplemental materials. Furthermore, emission factor values for tyre and brake wear reduce progressively from urban to rural to motorway roads, while the percentage increase of tyre wear emissions is equal, and the percentage increase of brake wear emissions rises gradually. It is also seen in Table 3 that the percentage increase of resuspension of road dust is the largest among the non-exhaust emissions, followed by road wear, brake wear, and tyre wear emissions.

Table 3. Increment (increased percentage) in non-exhaust emission factor (EF) from ICE petrol and diesel cars to their equivalent EVs (Unit: mg km^{-1} veh^{-1}).

Non-exhaust emissions	ICE cars switching to equivalent EVs	Emission factors	Urban roads	Rural roads	Motorway roads
Tyre wear	Petrol car to	$EF_{PM2.5}$	0.63 (9.73%)	0.49 (9.73%)	0.41 (9.73%)
	equivalent EV	EF_{PM10}	0.89 (9.73%)	0.70 (9.73%)	0.60 (9.73%)
	Diesel car to	$EF_{PM2.5}$	0.51 (7.33%)	0.39 (7.33%)	0.33 (7.33%)
	equivalent EV	EF_{PM10}	0.71 (7.33%)	0.56 (7.33%)	0.48 (7.33%)
Brake wear	Petrol car to	$EF_{PM2.5}$	0.58 (11.45%)	0.33 (14.72%)	0.09 (17.17%)
	equivalent EV	EF_{PM10}	1.51 (11.45%)	0.83 (14.72%)	0.22 (17.17%)
	Diesel car to	$EF_{PM2.5}$	0.46 (8.94%)	0.27 (11.46%)	0.07 (13.33%)
	equivalent EV	EF_{PM10}	1.21 (8.94%)	0.67 (11.46%)	0.18 (13.33%)
			Urban/Rural/Motorway roads		
Road wear	Petrol car to	$EF_{PM2.5}$	0.52 (14.72%)		
	equivalent EV	EF_{PM10}	0.94 (14.72%)		
	Diesel car to	$EF_{PM2.5}$	0.42 (11.46%)		

	equivalent EV	EF_{PM10}	0.76 (11.46%)
Resuspension	Petrol car to equivalent EV	$EF_{PM2.5}$	0.56 (20.59%)
		EF_{PM10}	2.31 (20.59%)
	Diesel car to equivalent EV	$EF_{PM2.5}$	0.46 (15.94%)
		EF_{PM10}	1.88 (15.94%)

To achieve a longer cruising range, EVs require larger batteries and more structural weight to accommodate them, which inevitably increases non-exhaust emissions and thus raises percentage increase in emission factors for the move from ICEVs to EVs (Shiau et al., 2009). Table 4 lists the increased percentage in non-exhaust EFs when petrol cars increase in mass by 10% and 50%. It can be seen that with the increase in vehicle weight, the suspension of road dust and tyre wear EFs have the maximum and minimum growth, respectively. In the case of brake wear EFs, the increased percentage reduces gradually from urban to rural to motorway roads.

Table 4. Increment (increased percentage) in non-exhaust emission factor (EFs) when the petrol cars increase in mass by 10% and 50% (Unit: mg km^{-1} veh^{-1}).

Non-exhaust emissions	Increased percentage in petrol car weight	Emission factors	Urban roads	Rural roads	Motorway roads
Tyre wear	10%	$EF_{PM2.5}$	0.28 (4.22%)	0.22 (4.22%)	0.19 (4.22%)
		EF_{PM10}	0.40 (4.22%)	0.31 (4.22%)	0.27 (4.22%)
	50%	$EF_{PM2.5}$	1.30 (19.28%)	1.01 (19.28%)	0.85 (19.28%)
		EF_{PM10}	1.84 (19.28%)	1.43 (19.28%)	1.23 (19.28%)
Brake wear	10%	$EF_{PM2.5}$	0.26 (5.13%)	0.15 (6.54%)	0.04 (7.58%)
		EF_{PM10}	0.68 (5.13%)	0.37 (6.54%)	0.10 (7.58%)
	50%	$EF_{PM2.5}$	1.20 (23.79%)	0.70 (31.04%)	0.19 (36.60%)
		EF_{PM10}	3.14 (23.79%)	1.76 (31.04%)	0.48 (36.60%)
			Urban/Rural/Motorway roads		
Road wear	10%	$EF_{PM2.5}$	0.23 (6.54%)		
		EF_{PM10}	0.42 (6.54%)		
	50%	$EF_{PM2.5}$	1.09 (31.04%)		
		EF_{PM10}	1.99 (31.04%)		
Resuspension	10%	$EF_{PM2.5}$	0.25 (9.02%)		
		EF_{PM10}	1.01 (9.02%)		
	50%	$EF_{PM2.5}$	1.22 (44.57%)		
		EF_{PM10}	5.00 (44.57%)		

3.1.2. Exhaust emission factors

To make a complete estimation of the effect of ICE passenger cars switching to their equivalent EVs, the emission factors from the tailpipe pollutants of ICEVs are also required. Here, emission factors from Euro 6 engine used in the UK National Atmospheric Emissions Inventory (Brown et al., 2018) are employed and summarised in Table 5. The ratio of PM_{10} to $PM_{2.5}$ in this inventory is 1.0, which implies that all exhaust particle emissions are in the range of $PM_{2.5}$ size. Compared to the $PM_{2.5}$ emission factor from diesel cars, petrol cars have lower values on urban roads and higher values on rural and motorway roads. NO_x and VOC emissions from petrol cars on urban, rural, and motorway roads are higher than those for diesel cars, whereas NH_3 emissions are lower from petrol cars.

Table 5. Exhaust emission factors emitted from Euro 6 petrol and diesel cars (Brown et al., 2018)

Vehicle type	Exhaust emission factors (mg km^{-1} veh^{-1})	Urban roads	Rural roads	Motorway roads
Petrol passenger car	$PM_{2.5}$	1.46	1.24	1.80
	NO_x	28.6	19.0	11.5
	VOC	3.53	3.12	4.66
	NH_3	4.1	8.0	21.8
Diesel passenger car	$PM_{2.5}$	1.49	1.11	0.90
	NO_x	44.9	39.5	52.5
	VOC	7.16	5.61	3.99
	NH_3	1.9	1.9	1.9

3.2. Monetary impact values of exhaust and non-exhaust emissions

3.2.1. Monetary impact values of non-exhaust emissions

The results of mean monetary impact values, alongside the low and high sensitivities for the non-exhaust particle emissions from ICEVs and their equivalent EVs, according to the non-exhaust emission factors and corresponding damage costs, are summarised in Table 6. The mean monetary impact values on three road conditions are in the range of 3.51×10^{-4} – 6.03×10^{-4} £ km^{-1} veh^{-1} for tyre wear, 4.25×10^{-5}–8.10×10^{-4} £ km^{-1} veh^{-1} for brake wear, 2.87×10^{-4}–4.06×10^{-4} £ km^{-1} veh^{-1} for road wear, and 2.23×10^{-4}–7.84×10^{-4} £ km^{-1} veh^{-1} for road dust resuspension. Close inspection of Fig. 2 and Table 6 shows that although PM2.5 emissions from tyre wear are less than PM_{10} emissions, the monetary impact values for $PM_{2.5}$ emissions are higher than those for PM_{10} emissions. This phenomenon is mainly because $PM_{2.5}$ emissions have higher damage costs compared to PM_{10} emissions. Amato et al., (2014) summarised that the smaller size of PM emissions caused the high deposition efficiency, which raised the possibility of toxicity effects to the lung and the penetration into the blood stream. Deng et al., (2019) revealed that compared to PM_{10}, $PM_{2.5}$ was more likely to deposit on and travelled into the surface of the lung, which induced tissue damage and lung inflammation, having severe damage costs.

As for all non-exhaust particle emissions, the $PM_{2.5}$ generated from brake wear of petrol vehicles on motorway roads have the smallest monetary impact values due to low-frequency braking times, while the PM_{10} from brake wear of diesel-equivalent EVs on urban roads possesses the largest monetary impact values. As expected, the monetary impact values of tyre and brake wear emissions for ICEVs and EVs on urban roads are larger than those on rural and motorway roads. Such differences in monetary impact values are closely associated with more emissions generated due to the higher frequency of acceleration and deceleration manoeuvres on urban roads compared to rural and motorway roads. Beji et al., (2020) studied non-exhaust emissions under various driving conditions and discovered a sharp increase in tyre and road wear particles during acceleration and deceleration phases.

Table 6. Mean monetary impact values (IVs), alongside low and high sensitivities (Min IVs and Max IVs) of non-exhaust emissions (eq-equivalent, Unit: £ km^{-1} veh^{-1}).

Tyre wear			Urban roads			Rural roads			Motorway roads		
			Mean IVs	Min IVs	Max IVs	Mean IVs	Min IVs	Max IVs	Mean IVs	Min IVs	Max IVs
ICEVs	Petrol	$PM_{2.5}$	5.49E-4	1.18E-4	1.70E-3	4.26E-4	9.19E-5	1.32E-3	3.60E-4	7.76E-5	1.12E-3
		PM_{10}	5.23E-4	1.13E-4	1.62E-3	4.08E-4	8.79E-5	1.26E-3	3.51E-4	7.56E-5	1.09E-3
	Diesel	$PM_{2.5}$	5.62E-4	1.21E-4	1.74E-3	4.36E-4	9.40E-5	1.35E-3	3.68E-4	7.93E-5	1.14E-3
		PM_{10}	5.35E-4	1.15E-4	1.66E-3	4.17E-4	8.99E-5	1.29E-3	3.59E-4	7.73E-5	1.11E-3
EVs	Petrol-eq	$PM_{2.5}$	6.01E-4	1.29E-4	1.86E-3	4.66E-4	1.00E-4	1.45E-3	3.94E-4	8.48E-5	1.22E-3
		PM_{10}	5.72E-4	1.23E-4	1.77E-3	4.46E-4	9.62E-5	1.38E-3	3.83E-4	8.26E-5	1.19E-3
	Diesel-eq	$PM_{2.5}$	6.03E-4	1.30E-4	1.87E-3	4.68E-4	1.01E-4	1.45E-3	3.95E-4	8.52E-5	1.22E-3
		PM_{10}	5.74E-4	1.24E-4	1.78E-3	4.48E-4	9.65E-5	1.39E-3	3.85E-4	8.30E-5	1.19E-3
Brake wear			Mean IVs	Min IVs	Max IVs	Mean IVs	Min IVs	Max IVs	Mean IVs	Min IVs	Max IVs
ICEVs	Petrol	$PM_{2.5}$	4.11E-4	8.85E-5	1.27E-3	1.85E-4	3.98E-5	5.73E-4	4.25E-5	9.16E-6	1.32E-4
		PM_{10}	7.24E-4	1.56E-4	2.24E-3	3.11E-4	6.70E-5	9.63E-4	7.16E-5	1.54E-5	2.22E-4
	Diesel	$PM_{2.5}$	4.22E-4	9.09E-5	1.31E-3	1.91E-4	4.12E-5	5.93E-4	4.43E-5	9.54E-6	1.37E-4
		PM_{10}	7.44E-4	1.60E-4	2.31E-3	3.22E-4	6.93E-5	9.97E-4	7.45E-5	1.60E-5	2.31E-4
EVs	Petrol-eq	$PM_{2.5}$	4.58E-4	9.86E-5	1.42E-3	2.12E-4	4.57E-5	6.57E-4	4.98E-5	1.07E-5	1.54E-4
		PM_{10}	8.07E-4	1.74E-4	2.50E-3	3.57E-4	7.68E-5	1.11E-3	8.38E-5	1.81E-5	2.60E-4
	Diesel-eq	$PM_{2.5}$	4.60E-4	9.91E-5	1.43E-3	2.13E-4	4.59E-5	6.61E-4	5.02E-5	1.08E-5	1.55E-4
		PM_{10}	8.10E-4	1.75E-4	2.51E-3	3.59E-4	7.73E-5	1.11E-3	8.44E-5	1.82E-5	2.62E-4

Road wear			Urban/Rural/Motorway roads		
			Mean IVs	Min IVs	Max IVs
ICEVs	Petrol	$PM_{2.5}$	2.87E-4	6.19E-5	8.91E-4
		PM_{10}	3.52E-4	7.59E-5	1.09E-3
	Diesel	$PM_{2.5}$	2.97E-4	6.41E-5	9.22E-4
		PM_{10}	3.65E-4	7.86E-5	1.13E-3
EVs	Petrol-eq	$PM_{2.5}$	3.30E-4	7.10E-5	1.02E-3
		PM_{10}	4.04E-4	8.71E-5	1.25E-3
	Diesel-eq	$PM_{2.5}$	3.32E-4	7.14E-5	1.03E-3
		PM_{10}	4.06E-4	8.76E-5	1.26E-3
Resuspension dust			Mean IVs	Min IVs	Max IVs
ICEVs	Petrol	$PM_{2.5}$	2.23E-4	4.81E-5	6.92E-4
		PM_{10}	6.16E-4	1.33E-4	1.91E-3
	Diesel	$PM_{2.5}$	2.34E-4	5.04E-5	7.25E-4
		PM_{10}	6.46E-4	1.39E-4	2.00E-3
EVs	Petrol-eq	$PM_{2.5}$	2.69E-4	5.80E-5	8.34E-4
		PM_{10}	7.43E-4	1.60E-4	2.30E-3
	Diesel-eq	$PM_{2.5}$	2.71E-4	5.85E-5	8.41E-4
		PM_{10}	7.48E-4	1.61E-4	2.32E-3

3.2.2. Monetary impact values of exhaust emissions

Table 7 summarises the mean monetary impact values, alongside the low and high sensitivities, of the exhaust emissions from ICEVs on urban, rural and motorway roads. It can be seen that the mean monetary impact values of $PM_{2.5}$, NO_x, VOC, and NH_3 generated from ICEVs on three road conditions are in the range of 7.34×10^{-5}–1.47×10^{-4} £ km^{-1} veh^{-1}, 1.04×10^{-4}–4.76×10^{-4} £ km^{-1} veh^{-1}, 3.18×10^{-7}–7.30×10^{-7} £ km^{-1} veh^{-1}, and 1.51×10^{-5}–1.73×10^{-4} £ km^{-1} veh^{-1}, respectively. Compared to exhaust emissions of petrol cars, diesel cars present higher monetary impact values of NO_x and VOC on most road conditions, while the impact values of $PM_{2.5}$ and NH_3 generated from diesel cars are lower except for urban roads. The NO_x generated from diesel cars on motorway roads contributes to the largest mean monetary impact value of 4.76×10^{-4} £ km^{-1} veh^{-1}, while the mean monetary impact value of the VOC emitted from petrol cars on rural roads is lowest. The results derived from economic–environmental analyses are useful for designing exhaust emission standards of ICEVs. For instance, the impact factors of VOC emitted from ICEVs are clearly larger than those of $PM_{2.5}$, as illustrated in Table 5, but the monetary impact values of VOC are much lower compared to those of $PM_{2.5}$.

Table 7. Mean monetary impact values (IVs) for exhaust emissions, alongside low and high sensitivities (Unit: £ km^{-1} veh^{-1}).

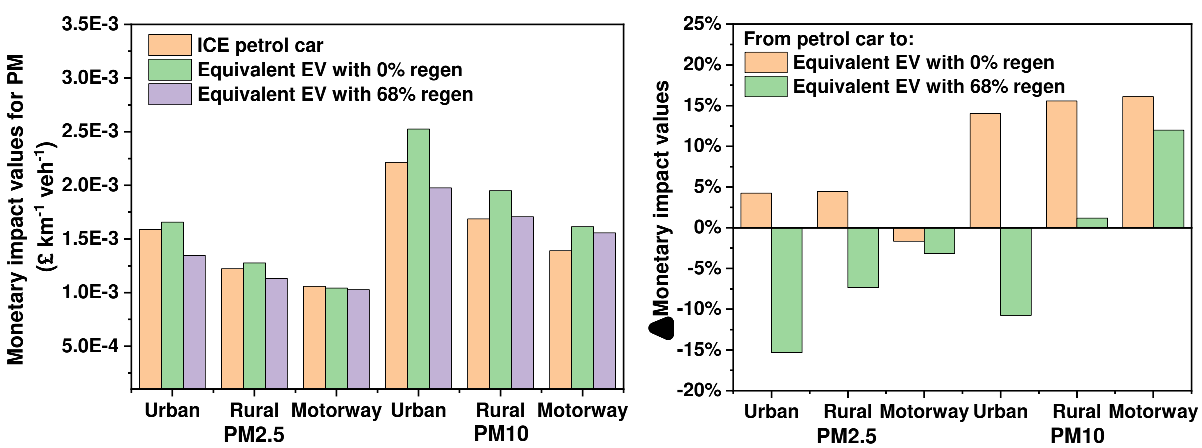
			Urban roads			Rural roads			Motorway roads		
		Exhaust emissions	Mean IVs	Min IVs	Max IVs	Mean IVs	Min IVs	Max IVs	Mean IVs	Min IVs	Max IVs
ICEVs	Petrol	$PM_{2.5}$	1.19E-4	2.56E-5	4.55E-4	1.01E-4	2.18E-5	3.13E-4	1.47E-4	3.16E-5	4.55E-4
		NO_x	2.59E-4	2.34E-5	4.00E-4	1.72E-4	1.55E-5	6.60E-4	1.04E-4	9.40E-6	4.00E-4
		VOC	3.60E-7	1.94E-7	9.55E-7	3.18E-7	1.72E-7	6.40E-7	4.75E-7	2.56E-7	9.55E-7
		NH_3	3.25E-5	6.24E-6	5.34E-4	6.34E-5	1.22E-5	1.96E-4	1.73E-4	3.32E-5	5.34E-4
	Diesel	$PM_{2.5}$	1.21E-4	2.62E-5	2.27E-4	9.05E-5	1.95E-5	2.80E-4	7.34E-5	1.58E-5	2.27E-4
		NO_x	4.07E-4	3.67E-5	1.82E-3	3.58E-4	3.23E-5	1.37E-3	4.76E-4	4.29E-5	1.82E-3
		VOC	7.30E-7	3.94E-7	1.47E-6	5.72E-7	3.09E-7	1.15E-6	4.07E-7	2.19E-7	8.18E-7
		NH_3	1.51E-5	2.89E-6	4.65E-5	1.51E-5	2.89E-6	4.65E-5	1.51E-5	2.89E-6	4.65E-5

3.3. Comparative analysis of PM monetary impact values

The total PM monetary impact values of ICE passenger cars and the equivalent EVs with 0% and 68% regenerative braking were calculated according to PM's mean damage values and the corresponding emission factors. Fig. 3 shows the total PM monetary impact values and percentage variation in PM monetary impact values from conventional cars to corresponding EVs. Without considering regenerative braking, the $PM_{2.5}$ impact values of equivalent EVs are 1.76–3.04% larger than those of diesel cars on urban, rural, and motorway roads and 4.26–4.44% larger than those of petrol cars on urban and rural roads. With regard to PM_{10} monetary impact values, the equivalent EVs without regenerative braking on urban, rural and motorway roads have 14.02–16.09% and 10.94–12.53% larger monetary impact values than petrol and diesel cars, respectively. This phenomenon is primarily because the non-exhaust PM emissions are highly associated with vehicle weight (Kakad et al., 2017; Rajamani et al., 2010; Timmers and Achten 2016). Compared to ICE petrol cars, the equivalent EVs have heavier vehicle weight (see Table S1 of the supplemental materials), which emits more non-exhaust emissions, hence causing higher total monetary values of PM emissions. In addition, it can be observed that from Fig. 3, most of Δ monetary impact values for the switch to EVs with 68% regenerative braking from ICEVs are negative. This is ascribed to the fact that EVs with 68% regenerative braking can reduce 68% brake wear $PM_{2.5}$ and PM_{10} emissions, thus contributing to lower total monetary impact values of PM emissions.

Emission factors and damage cost values of PM_{10} and $PM_{2.5}$ have great impacts on the monetary impact values. In the report of the Organization for Economic Co-operation and Development (OECD) (OECD, 2020), it was found that battery electric vehicles with a 100-mile range (EV-100) emitted 17.8% and 12.8% less PM_{10} and $PM_{2.5}$ emissions than their conventional gasoline-fueled passenger cars. Battery electric vehicles with a 300-mile range (EV-300) reduced PM_{10} emissions by 6.5% and increased $PM_{2.5}$ emissions by 2.6% relative to conventional passenger cars. These results seem to be contrary to our current results under certain conditions. Such a difference can be attributed to the different assumptions underlying the estimation of non-exhaust emissions. Firstly, we use a different reduction value in braking wear PM_{10} and $PM_{2.5}$ emissions. The OECD assumed a 75% reduction in brake wear PM_{10} and $PM_{2.5}$ emissions for EV-100 and EV-300 from the regenerative braking system (OECD, 2020). In contrast, our research assumes 0% and 68% reductions in brake wear PM_{10} and $PM_{2.5}$ emissions due to regenerative braking. The EVs with 68% regenerative braking are chosen in our work according to the average values of reduced emission due to regenerative braking in the published papers (see Fig. 1). Secondly, non-exhaust PM_{10} and $PM_{2.5}$ emissions in multiple scenarios, such as urban, rural and motorway roads, are considered in the current work, whereas this is not emphasized by the OECD report. Thirdly, we take into account vehicle weight in modelling PM emissions. In the report of OECD (OECD, 2020), although the vehicle weight of EV-100 and EV-300 was larger than ICEV, PM_{10} and $PM_{2.5}$ emissions of road wear and road dust resuspension were equally assumed for ICEV, EV-100 and EV-300. This assumption is different from extensive studies, which have revealed that an increase in vehicle weight increases road wear and road dust resuspension emissions (Beddows and Harrison, 2021; Gillies et al., 2005; Piscitello et al., 2021; Simons, 2013; Singh et al., 2020; Timmers and Achten, 2016, 2018; U.S. EPA, 2011). In light of this, we consider vehicle weight as a key factor affecting road wear and road dust resuspension emissions in our research and apply the method proposed by Beddows and Harrison (2021) to capture the role of vehicle weight in modelling emissions. In summary, the discussion above could be the main reasons for the difference in the results of our research and the report of OECD. Meanwhile, we reviewed some relevant literature and found that whether the conversion of ICEV to EV reduces PM emissions may highly depend on specific study conditions. For instance, a recent review by Requia et al. (2018) evaluated 4734 studies regarding the impact of the move from ICEVs to EVs on environmental aspects. 65 studies fulfilled the inclusion criteria for the review, and the results consistently revealed reductions in greenhouse gas emissions and emissions of some criteria pollutants, while the increase or decrease in PM depended strongly upon the study conditions. Beddows and Harrison (2021) identified that non-exhaust PM_{10} and $PM_{2.5}$ emissions from EVs without a regenerative braking system would exceed PM_{10} and $PM_{2.5}$ emissions, including exhaust, from ICEVs. Huo et al. (2015) et al. assessed the life-cycle emissions of EVs and ICEVs in three Chinese regions and three U.S. regions in 2012. The obtained results showed that compared to ICEVs, EVs in all regions except the Northeast Power Coordinating Council in the U.S. would increase PM_{10} and $PM_{2.5}$ emissions. Timmers and Achten (2016) pointed out that PM_{10} emissions from EVs were nearly equal to those from ICEVs, and $PM_{2.5}$ emissions from EVs were 3% lower than those from ICEVs.

In the present study, it is found that the regenerative braking system is a key factor affecting whether the conversion of ICEV to EV reduces total PM monetary impact values. The total PM_{10} monetary impact values are increased for the conversion of ICEVs to EVs without regenerative braking. As a result, regenerative braking of EVs is required to decrease brake wear emissions to make total monetary impact values equal to those of equivalent ICEVs. In line with the PM_{10} monetary impact values of petrol cars, the equivalent EVs need to reach 38.49% regenerative braking on urban roads and 73.65% regenerative braking on rural roads. In comparison, diesel-equivalent EVs require less extent of regenerative braking. For example, 30.90% and 59.16% regenerative braking are needed on urban and rural roads to be consistent with the PM_{10} monetary impact values of diesel cars. On motorway roads, however, the PM_{10} monetary impact values of equivalent EVs, even with 100% regenerative braking, are still higher than those of the corresponding petrol and diesel cars. Compared to the total PM_{10} monetary impact values, the total $PM_{2.5}$ monetary impact values of equivalent EVs without regenerative braking present smaller increases compared to those of the ICEVs, as illustrated in Fig. 3. Therefore, a smaller degree of regenerative braking is required to be consistent with the $PM_{2.5}$ monetary impact values of petrol and diesel cars. Petrol-equivalent EVs with 14.79% and 25.61% are needed on urban and rural roads, while there is no requirement of regenerative braking on motorway roads. For diesel-equivalent EVs, it is required to have 6.27%, 16.31%, and 61.61% regenerative braking on urban, rural and motorway roads, respectively.



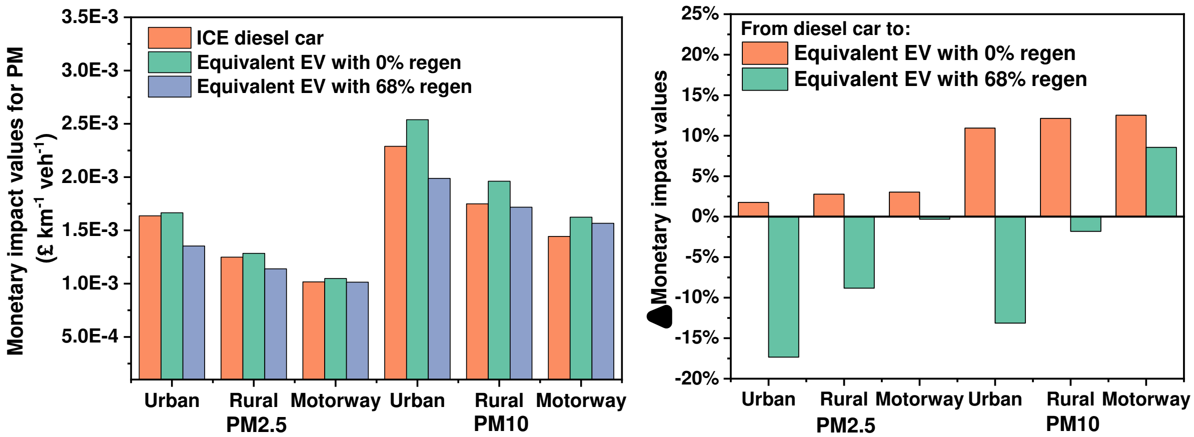


Fig. 3. The left panels show monetary impact values of total particulate matter (PM) emitted from ICE petrol or diesel cars and equivalent EVs. The right panels present the percentage variation in monetary impact values from ICE petrol or diesel cars to equivalent EVs with 0% or 68% regenerative braking.

3.4. Comparative analysis of emission monetary impact values

Fig. 4 presents total monetary impact values of the emissions from petrol or diesel passenger cars and their equivalent EVs with 0% and 68% regenerative braking on urban, rural and motorway roads and percentage variation in total monetary impact values of emissions from the conversion of conventional cars to their equivalent EVs. The average monetary impact values of total emissions from petrol cars and their equivalent EVs with 0% regenerative braking are 4.10×10^{-3} and 4.18×10^{-3}£ km^{-1} veh^{-1} on urban roads, 3.15×10^{-3} and 3.23×10^{-3}£ km^{-1} veh^{-1} on rural roads, and 2.73×10^{-3} and 2.66×10^{-3}£ km^{-1} veh^{-1} on motorway roads, respectively. Compared to petrol passenger cars and their equivalent EVs, diesel passenger cars and their equivalent EVs exhibit slightly higher monetary impact values.

Without considering regenerative braking, the percentage increases in total monetary impact values of emissions from the conversion of petrol cars to equivalent EVs are 2.10% on urban roads and 2.57% on rural roads. In the study by Liang et al., (2019), however, they indicated that fleet electrification in China would deliver greater air quality, climate and health benefits. This conclusion seems to be against our current results, which is primarily since PM_{10} emissions were not considered in this published paper. It can be seen that from Fig. 3, the monetary impact values for PM_{10} from EVs on urban and rural roads without regenerative braking are apparently higher than those from ICE petrol passenger cars. Thus, this would inevitably lead to an increase in total monetary impact values. It is worth mentioning that the percentage increase in total monetary impact values of emissions for the move from petrol cars to equivalent EVs without regenerative braking is -2.61% on motorway roads. From Tables 1 and 5, it can be seen that petrol cars on motorway roads emit much more NH_3 and exhaust $PM_{2.5}$ emissions than on urban and rural roads, and these two emissions have large damage cost values. Thus, they contribute to higher total monetary impact values for petrol passenger cars on motorway roads than those for equivalent EVs with 0% regenerative braking. For the conversion of diesel cars to their equivalent EVs, the total monetary impact values of emissions reduce on urban, rural and motorway roads. Although the equivalent EVs with 68% regenerative braking contribute to less monetary impact values than the ICEVs, it should be noted that regenerative braking can be used for ICEVs to reduce the particles emitted from brake wear (Hall, 2017). In addition, it is worth mentioning that if the electricity needed to fuel the battery is not produced by photovoltaic and wind power plants, the monetary impact values of pollutants emitted during the electricity production must also be considered.

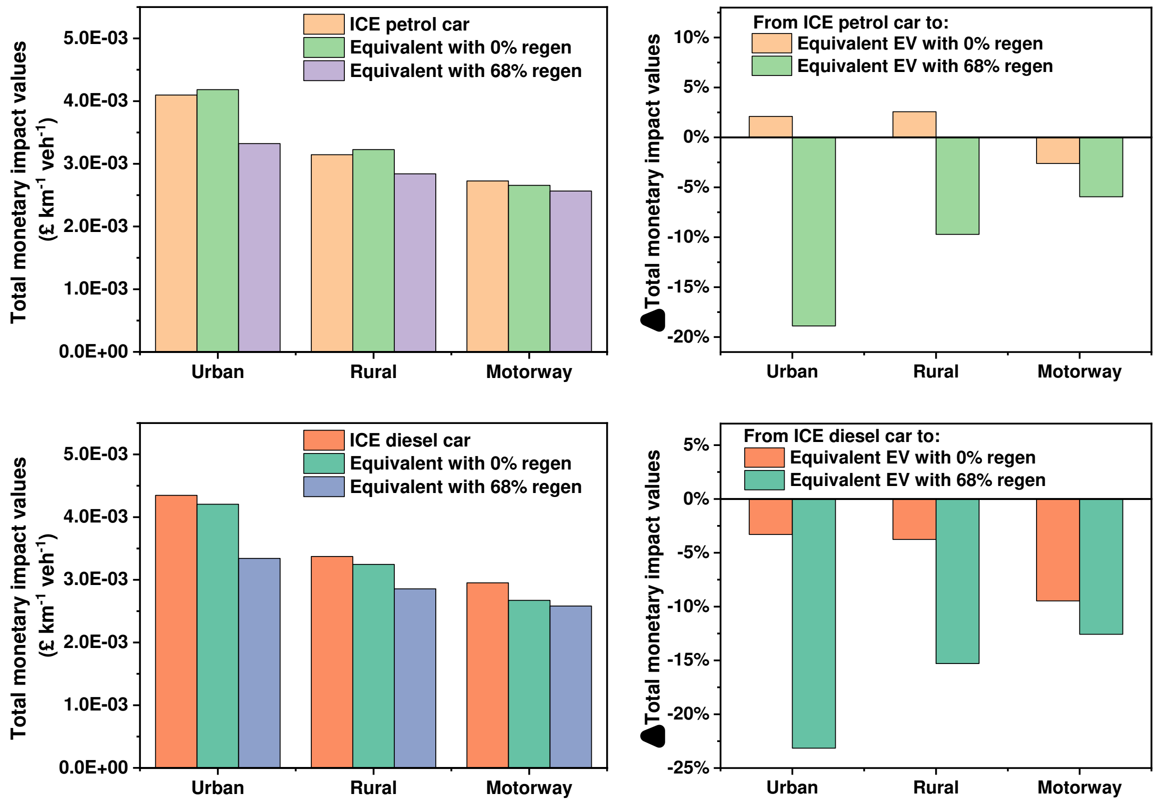


Fig. 4. The left panels present total monetary impact values of the emissions from ICE petrol or diesel passenger cars and their equivalent EVs with 0% and 68% regenerative braking on urban, rural and motorway roads. The right panels show the percentage variation in total monetary impact values of emissions from ICE petrol or diesel cars to their equivalent EVs with 0% and 68% regenerative braking.

This study is the first attempt to monetise the exhaust and non-exhaust emissions emitted from EVs and ICEVs. In the current work, only monetary impact values of the emitted emissions from the conventional passenger cars and their equivalent EVs are calculated since the statistics of these vehicle weights are available to obtain emission factors. When the emission factors of other categories of vehicles and damage costs of these emissions are available, the monetary impact values of these vehicles will be evaluated to provide more comprehensive references for the regulatory authorities and policy makers to design the mitigation strategies. In addition, the conventional vehicles would generate non-regular emissions, such as aldehydes and ketones. If the damage cost values of these emissions are known, they should be calculated to obtain total monetary impact values from conventional vehicles. In the last couple of years, research into non-exhaust emissions emitted from ICEVs and EVs has increased, but more experimental data, especially from EVs, are needed to get a comprehensive understanding of the monetary impact values, which is beneficial for the regulatory authorities and policymakers to design the mitigation strategies and to compute their contributions and impacts on public health and local air quality. Based on the current data, further improvements need to be made by encouraging innovation in reducing vehicle weight since vehicle weight significantly affects non-exhaust emissions and monetary impact values. In addition, regenerative braking can effectively reduce brake wear emissions, especially on urban roads. Thus, the government should create incentives for consumers and car manufacturers to equip with a regenerative braking system for more vehicles and switch to more lightweight passenger cars.

4. Conclusions

In the present study, we have proposed the methodology to assess the monetary impact values of exhaust and non-exhaust emissions from ICEVs and their equivalent EVs. We have also compared the monetary impact values of these emissions to identify the effect of ICE passenger cars switching to their equivalent EVs. The current results show that the total monetary impact values decrease for the switch to equivalent EVs without regenerative braking from ICE diesel passenger cars. In the case of the switch to equivalent EVs without regenerative braking from ICE petrol passenger cars, however, the total monetary impact values of emissions increase on urban and rural roads. This is mainly because the monetary impact values of exhaust emissions from ICE petrol cars are smaller than those from ICE diesel cars, leading to relatively small total monetary impact values, which is less than those from equivalent EVs. The ICE passenger cars switching to equivalent EVs with 68% regenerative braking would reduce significantly total monetary impact values. Thus, regenerative braking should be encouraged to be installed to minimise brake wear emissions. These results can be useful for the economic-environmental evaluation of vehicle emissions. Since this work is the first attempt to calculate the monetary impact values of vehicle exhaust and non-exhaust emissions, the results are likely to be improved when new data regarding the emission factors and damage costs of vehicle exhaust and non-exhaust emissions are available.

5. CRediT authorship contribution statement

Ye Liu: Investigation, Methodology, Data visualisation, Writing-original draft. **Haibo Chen**: Conceptualisation, Investigation, Funding acquisition, Project management. **Ying Li**: Methodology, Writing-review & editing. **Jianbing Gao**: Methodology, Writing-review & editing. **Kaushali Dave**: Investigation, Writing-review & editing. **Junyan Chen**: Writing-review & editing. **Tiezhu Li:** Conceptualization, Writing-review & editing**. Ran Tu:** Writing-review & editing

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