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Environmental and health impacts of banning passenger cars with internal combustion engines: A case study of Leeds, UK

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ABSTRACT

Governments worldwide are considering banning the sale of internal combustion engine vehicles (ICEVs) to address environmental and public health issues. However, the environmental and health impacts of banning ICEV sales remain unclear. Here, we evaluated the effects of banning ICEV sales under various implementation timelines on $PM_{2.5}$ and NO_x emissions in Leeds, UK, and the associated premature deaths, using a novel 'Fleet-Emission-Health' model. The results showed that the earlier ban on ICEV sales led to an almost 100% reduction in NO_x by 2040, whereas total $PM_{2.5}$ increased slightly due to more non-exhaust $PM_{2.5}$ emissions emitted from battery electric vehicles (BEVs). Moreover, banning ICEV sales in 2030 would avoid 28 deaths in Leeds in 2040 compared to those in 2022, due to the reduction of NO_x and $PM_{2.5}$ emissions from ICEV exhaust. The findings indicate that stricter regulations on non-exhaust emissions are necessary to mitigate environmental and public health effects.

1. Introduction

Recently, there has been an increasing concern about the climate, environmental, and health impacts of pollutants from transport sources, particularly with regard to passenger cars with internal combustion engines. As a result, many countries around the world have proposed policies to ban the sales of internal combustion engine vehicles (ICEVs) in the near future (Senecal et al., 2021). For instance, Norway is a frontrunner, aiming to ban the sale of new ICEVs by 2025, while France and Spain plan to implement the ban on ICEV sales from 2040 onwards (Plötz et al., 2019). Other countries like Canada, Japan, and South Korea have also announced targets to end the sale of new ICEVs by 2040 or earlier (Senecal et al., 2021). The UK government initially announced a ban on the sales of new ICEVs from 2040 under the Road to Zero policy (UK government, 2018). This implementation time was subsequently revised to 2030 to align with the net-zero emissions target outlined in the Transport Decarbonisation Plan (UK government, 2021). However, the latest policy, incorporating the Zero Emission Vehicle (ZEV) Mandate, has postponed the implementation of this ban from 2030 to 2035 (UK government, 2023). Even though the actual implementation of the policy may differ from the initial announcement and plan, it is crucial to gain a thorough understanding in advance of the potential environmental and health implications associated with the

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timeline for the ban of ICEV sales.

ICEV ban would lead to a substantially reduced number of ICEVs and an increased number of battery electric vehicles (BEVs). Compared to ICEVs, the primary advantage of BEVs is the potential to achieve zero tailpipe emissions, including particulate matter (PM), nitrogen oxides (NO_x), carbon monoxide (CO), etc. Ferrero et al. (2016) discovered that the switch from ICEVs to BEVs could achieve a significant decrease in NO and NO_2 concentrations. Soret et al. (2014) reported that a 40 % electrification of vehicle fleets resulted in 11 % and 17 % reductions in total NO_x emissions in Barcelona and Madrid, respectively. Singh and Strømman (2013) pointed out that incorporating BEVs into passenger car fleets caused a reduction in emissions of NO_x, SO_2, and PM. It should be emphasized, however, that BEVs cannot be considered zero-emission vehicles. This is because BEVs emit non-exhaust PM from brake wear, tyre wear, road wear, and the resuspension of road dust (Harrison et al. 2021; Woo et al., 2022). It means that banning ICEV sales probably will not result in a substantial decrease in PM levels. Similarly, Timmers and Achten (2016) uncovered that BEVs would produce higher levels of non-exhaust emissions due to their heavier weight relative to ICEVs, and thus the transition to BEVs could not necessarily result in a substantial decrease in PM concentrations. Beddows and Harrison (2021) constructed a comprehensive model based on emission inventory to assess emission factors (EFs) of vehicle exhaust and non-exhaust PMs. The results showed that the transition from ICEVs to BEVs would emit more total PM_{10} and $PM_{2.5}$ emissions in most scenarios. Liu et al. (2021, 2022a) pointed out that non-exhaust PM from equivalent BEVs may exceed total PM from ICEV, including exhaust PM, which depended primarily upon the extent of regenerative braking, followed by the type of passenger car and road. Although the influence of vehicle electrification on exhaust and non-exhaust emissions has been reported in the literature, it remains unclear how banning ICEV sales under different schedules affects exhaust and non-exhaust emissions, and the significance of the environmental and health impacts as a result of the emissions.

In this context, an integrated modelling framework that takes into account fleet composition, environment, and human health was developed to assess the outcomes of enforcing the ban on ICEV sales under various timelines in Leeds, UK. The specific aspects include the city's demographic profile, dispersion patterns of pollutants, flow traffic, and fleet composition, which may differ in various cities. However, the methodological framework is broadly applicable and can be adapted for use in different contexts. In the current work, the following three sub-issues were formulated to be addressed: (1) Will the ban on ICEV sales result in $PM_{2.5}$ and NO_x reductions? And for how much? (2) How will the ban on ICEV sales change the share of emissions produced by each type of vehicle? (3) What are the health effects of the ban on ICEV sales?

2. Methodology

In the current work, three scenarios have been defined: the reference (REF), the ambitious ban policy (ABP), and the smooth ban policy (SBP). The summary of the three scenarios is shown in Table 1. To quantify the effects on the environment and human health, including annual NO_x emissions, exhaust and non-exhaust $PM_{2.5}$ emissions, emission harm equivalent (EHI), and premature deaths attributable to the emissions, a modelling framework was developed following the structure of 'Fleet-Emission-Health', as shown in Fig. 1. The essential steps involved in the modelling process include:

- Forecast the number of each vehicle class (BEV, plug-in hybrid electric vehicle (PHEV), and ICEV) without and with the ICEV ban policy leading to 2040.
- Determine the proportion of vehicles with various emission standards in the fleet across three scenarios.
- Calculate annual emissions of NO_x, exhaust and non-exhaust $PM_{2.5}$ under the three scenarios.
- Evaluate the EHI and the number of premature deaths attributable to NO_x and $PM_{2.5}$ emissions in Leeds under the three scenarios.

2.1. Fleet composition under REF scenario

A fleet turnover model was created to evaluate the fleet composition under the REF scenario between 2020 and 2040. The total number of each vehicle class was first calculated, then the age of the vehicles in the fleet was determined based on newly registered and scrapped vehicles for the calendar year, and finally, the proportion of vehicles equipped with different power drives and the proportion of ICEVs with different European emission standards were ascertained each year.

Table 1
Summary of three scenarios.

Scenario	Purpose	Year of Ban on ICEV Sales
REF	Examine the environmental and human health impacts based on the development of the existing vehicle electrification rates in the UK (Küfeoğlu and Hong, 2020).	Not applicable
ABP	Assess the environmental and human health impacts of banning ICVE sales starting in 2030	2030
SBP	Evaluate the environmental and health outcomes of enforcing a ban on ICEV sales beginning in 2035.	2035

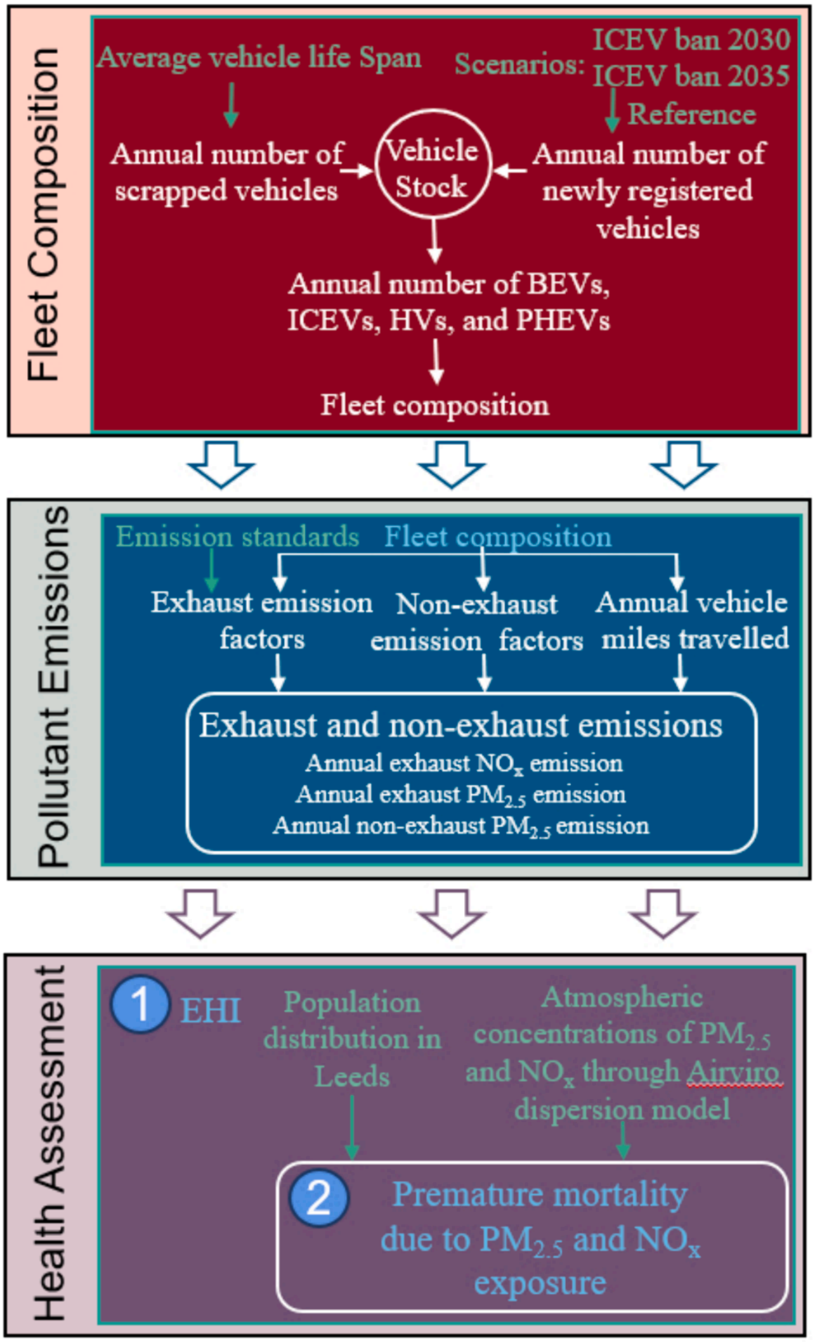

Fig. 1. The 'Fleet-Emission-Health' modelling framework.

2.1.1. The number of each vehicle class under REF scenario

The total vehicle number of each category in the UK without considering the sale ban policy on ICEVs was projected by the plan of the UK's transport sector (National grid, 2018; Küfeoğlu and Hong, 2020). The projected results are listed in Table S1 of the Supplementary Material. Additionally, it can be seen from Table S2 of Supplementary Material that the share of vehicles in Leeds to those in the UK was relatively stable. As a result, we used the mean proportion and the total quantity of vehicles in the UK to estimate the quantity of different types of vehicles in Leeds, as the REF scenario. The obtained results are illustrated in Fig. 2. As seen, the proportion of vehicles driven by electric and hybrid technologies is rising, while the proportion of vehicles powered by fossil fuels is declining.

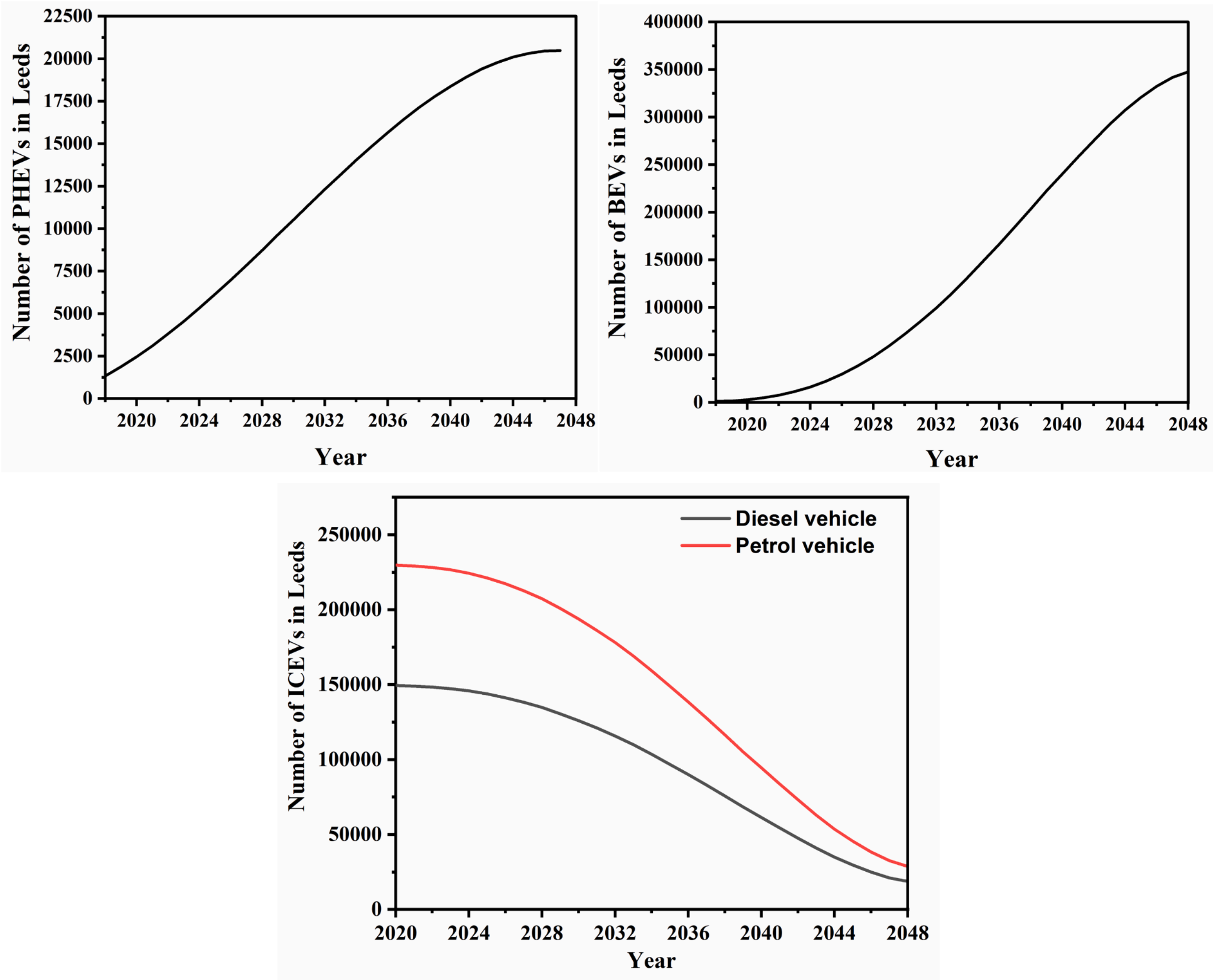

Fig. 2. The predicted number of each vehicle class in Leeds.

2.1.2. The number of vehicles with different European emission standards under REF scenario

While the number of each vehicle class under REF scenario has been obtained in section 2.1.1, the European emission standards implemented by vehicles in the fleet are not available. However, it is well known that the ICEVs complying with various emission standards emit different levels of NO_x and $PM_{2.5}$. To estimate NO_x and $PM_{2.5}$ emissions accurately, the number of scrapped and newly registered vehicles per year needs to be computed in order to determine the number of vehicles with different European emission standards.

The survival-age rate, denoting the annual percentage of vehicles that remained in the fleet (Mehlig et al., 2023), was employed to predict the number of scrapped vehicles. Previous studies have revealed that the relationship between vehicle age and survival rate generally conforms to the Weibull distribution (Brand et al., 2012; Craglia and Cullen, 2020a,b; Zachariadis et al., 1995; Zachariadis et al., 2001), expressed as (1):

$$\varphi(k) = \exp - [(\frac{k+b}{T})^b] \text{ and } \varphi(0) = 1 \tag{1}$$

where k denotes the age of the vehicle;$\varphi(k)$ represents the survival probability of the vehicle with age k; b represents the failure steepness; T represents the characteristic service life. The parameter values were set based on the number of registered and scrapped vehicles in the UK for 1995–2019, where $b = 6.5$ and T=22.5.

The number of all types of vehicles and respective ages in the fleet in 2021 can be determined according to public data from the UK government (UK Government, 2022). Based on the age of vehicles in the fleet in 2021 and the survival-age rates, the number of vehicles scrapped in 2021 can be derived. Equation (2) was then used to estimate the number of newly registered vehicles in 2022. By iterating this process, the age of vehicles in the fleet for 2022–2040 under REF scenario can be yielded. As a result, on the basis of the timetable for the implementation of the European emission standards, it is possible to obtain the proportion of ICEVs with different emission standards. Eq. (2) is shown as follows:

$$NV_{ik} = N_{ik} - \sum_{j \in J} R_{ijk} \tag{2}$$

where NV_{ik} refers to the quantity of newly registered vehicles with power-driven type i in year k under *REF* scenario;N_{ik} refers to the quantity of vehicles with power-driven type i in year k under *REF* scenario;R_{ijk} refers to the quantity of vehicles remaining under *REF* scenario with age j ($j \geq 1$) and power-driven type i in year k.

2.2. Fleet composition under ABP and SBP scenarios

In the ABP or SBP scenario, calculating fleet composition is the same as the REF scenario, except that the proportion of new registered vehicles is different from that in the REF scenario. To avoid the uptake rates of BEV sales from immediately jumping to 100 % on the date when the ban on ICEV sales takes effect, the logistic curve, also known as the S-shaped curve, was used to more plausibly predict the gradual change in the market share of BEVs before the implementation of the ban on ICEV sales (Tang et al., 2013). Following the implementation of the ban policy of ICEV sales, passenger vehicles sold in the market will be BEVs. On the basis of the determined market share of BEVs, the market shares of the other vehicle classes in the ABP and SBP scenarios can be determined. The obtained results are summarised in Table S3 of the Supplementary Material.

2.3. Vehicle mileage travelled

It is widely accepted that the mileage of a vehicle diminishes with vehicle age (Craglia and Cullen, 2020a,b). In the UK, the mileage-age rates for diesel and petrol vehicles have been evaluated based on the roadworthiness test of the UK Ministry of Transport (MOT) (Mehlig et al. 2023), and the determined results are shown in Fig. 3. In terms of BEVs, however, the lifecycle mileage-age rate has not yet been definitely established given their relatively recent adoption. The mileage-age rate of BEVs in the present work was assumed to be the average value of that for petrol and diesel vehicles, and the potential effects of the assumed value on vehicle emissions and human health are studied in a sensitivity analysis in Section 4.2. The vehicle mileage travelled for all classes of vehicles was calculated based on the known annual mileage driven for newly registered vehicles (Transport, 2022) UK government, 2023) and the mileage-age rates.

2.4. No_x and $PM_{2.5}$ emissions

NO_x is generated only from exhaust emissions of ICEVs, whereas $PM_{2.5}$ is emitted from both vehicle exhaust and non-exhaust emissions, which can be calculated from Eqs. (3) and (4), respectively:

$$E_n = \sum_{ijt} A_{ijt} \alpha_{ijn} \tag{3}$$

$$E_p = \sum_{ijt} A_{ijt} (\alpha_{ijp} + \beta_{ip}) \tag{4}$$

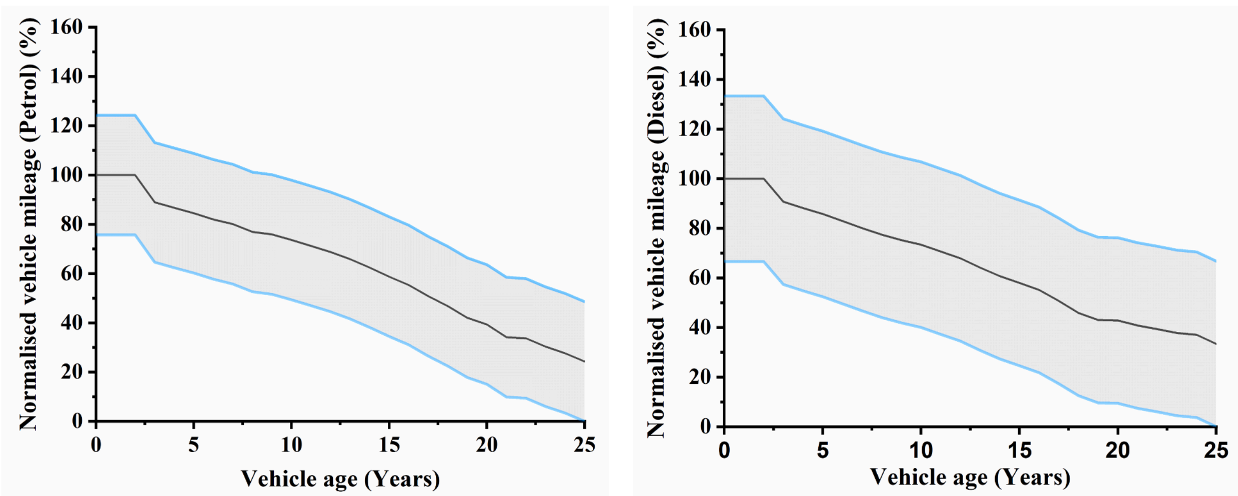

Fig. 3. Mileage-age rate distributions for petrol cars and diesel cars in the UK. The shaded areas denote the standard deviation range; the black line presents the mean mileage-age rate (Mehlig et al. 2023).

where E_n and E_p are the yearly emissions of NO_x and $PM_{2.5}$ in Leeds, respectively;A_{ijt} is the total annual vehicle mileage travelled of vehicles with power-driven type i, emission standard j, and vehicle age t in Leeds per year, which can be calculated according to fleet composition and mileage-age rate;α_{ijn} and α_{ijp} are EFs (see Table S5 of Supplementary material) for exhaust NO_x and $PM_{2.5}$, respectively, of vehicles with power-driven type i and emission standard j (Dallmann and Façanha, 2016; Lopes et al., 2014); and β_{ip} is the EF for non-exhaust $PM_{2.5}$ (see Fig. S4 of Supplementary material) for vehicles with power-driven type i, which was evaluated using the EMEP/EEA Guideline approach, with adjustments made for the impact of vehicle weight (Beddows and Harrison, 2021; Harrison et al., 2021; Liu et al., 2021; EU, 2023; EEA, 2016).

2.5. Health impact assessment

In the present work, two independent parameters, the emission harm equivalent index (EHI) and the number of premature deaths (Mylonakou et al., 2023), were employed to assess the public health impacts of traffic-related $PM_{2.5}$ and NO_x emissions. The EHI signifies the likelihood of an individual's death due to exposure to $PM_{2.5}$ and NO_x. The EHI is shown in Eq. (5):

$$EHI = \frac{1000\sum_m Eq_m\left(\sum_{ijtm} A_{ijt}\beta_{im} + \sum_{ijtm} A_{ijt}\alpha_{ijm}\right)}{cap} \tag{5}$$

where EHI is the emission harm equivalent index; The conversion factor,Eeq_m, quantifies the relative effects of pollutant type m in the UK on premature mortality due to exposure to $PM_{2.5}$, which was used to determine the relative EHI value of NO_x emissions. Here, the $Eeq_{NO_x} = 0.044$ was used based on the Thematic Strategy on Air Pollution's report (Amann, 2014), which focuses only on the health impacts of exposure to secondary inorganic aerosol (SIA) $PM_{2.5}$ resulting from NO_x emissions that do not include NO_2, without considering other health and vegetation impacts; β_{im} is non-exhaust EFs for pollutant m of vehicles with power-driven type i; α_{ijm} is EFs for pollutant m of vehicles with power-driven type i and emission standard j; cap denotes the number of people in Leeds which can be obtained from the database of the Leeds Population Laboratory (Leeds City Council: Leeds population laboratory. The Office of National Statistics (ONS), 2023).

In addition, the number of annual premature deaths attributable to atmospheric $PM_{2.5}$ and NO_x exposure was evaluated separately using the EEA's methodology (EEA, 2016). The calculation process mainly includes the following steps:

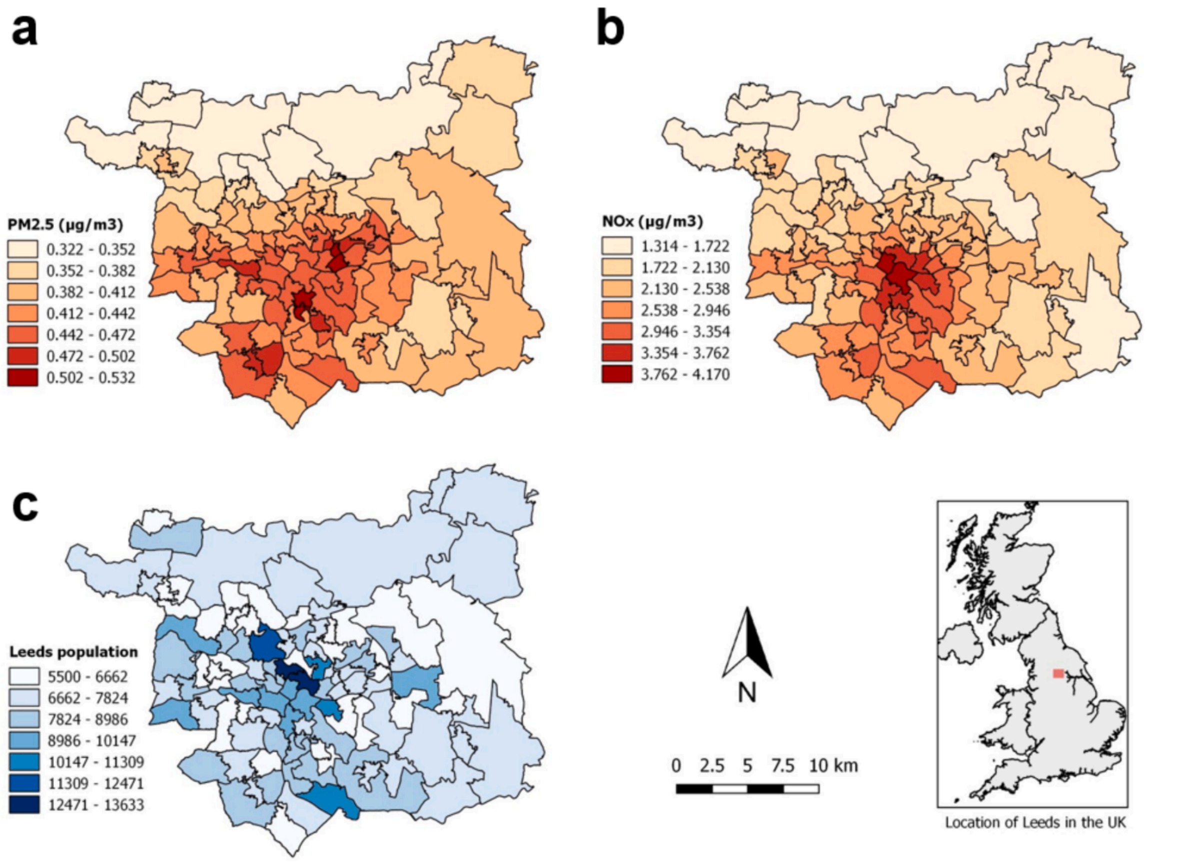

Fig. 4. Atmospheric concentrations of $PM_{2.5}$ and NO_x emissions in 2024 in the REF scenario and population in Leeds based on the 2021 census.

- Determination of the annual $PM_{2.5}$ and NO_x emissions from passenger cars in Leeds.
- A separate assessment of atmospheric concentration spatial maps of the $PM_{2.5}$ and NO_x emissions from passenger cars. The Airviro is an integrated system including emission inventories (Emission Data Base, EDB), dispersion modelling and data handling. In this study, the Gaussian dispersion model in the Airviro system was used to generate atmospheric concentration maps of $PM_{2.5}$ and NO_x emissions with a resolution of 1×1 km^2. Further details regarding the Airviro system can be found in the Supplementary Material. A case result on the mean concentration of $PM_{2.5}$ and NO_x emissions in Middle layer Super Output Areas (MSOAs) in 2024 is shown in Fig. 4. MSOA is the England census geographic unit, which usually comprises between 5,000 to 15,000 residents in each area.
- Calculation of premature deaths in each MSOA due to exposure to $PM_{2.5}$ and NO_x emissions. The equation for premature deaths recommended by the World Health Organization (WHO) was used in the current work (Leeuw et al. 2016):

$$P_{di} = AF_i \bullet MR \bullet Pop \tag{6}$$

Where P_d is the number of all-cause mortality attributable to the long-term exposure of pollutant i ($PM_{2.5}$ or NO_x); *Pop* is the size of the population (number of residents in each MSOA); *MR* is the baseline incidence of premature deaths (e.g. the total number of premature deaths per 100,000 people per year), which is from the statistical data of premature deaths from UK Office for National Statistics (ONS, 2022); AF_i is the attributable fraction of mortality from air pollution i for the exposed population, which is calculated using the Eq. (7):

$$AF_i = (RR_i - 1)/RR_i \tag{7}$$

$$RR_i = \exp(\beta \bullet (C - C_0)) \tag{8}$$

Where RR_i refers to the relative risks that capture the increase in mortality attributable to a given increase value in the air pollutant i concentration (i.e. concentration–response functions). The concentration–response functions defined by the WHO (EEA, 2016) were employed in this study; C and C_0 refer to simulated concentrations of pollutant i with and without considering emissions from passenger cars, respectively; β is the concentration–response factor, which is determined based on the UK official report (UK Government, 2014; Walton et al., 2015). For instance, the relative risk is 1.06 (95 % Confidence Interval (CI) 1.04–1.08) per 10 μg/m^3 increase in $PM_{2.5}$ and 1.039 (95 % CI 1.022, 1.056) per 10 μg/m^3 increase in NO_x, respectively.

3. Results and discussion

3.1. No_x emissions

Fig. 5 illustrates annual NO_x emissions under the REF, STP, and ABP scenarios. As expected, the introduction of a ban on ICEV sales resulted in a significant reduction in annual NO_x emissions. More precisely, under the SBP and ABP scenarios, annual NO_x emissions in Leeds were projected to decrease from 413 tonnes and 403 tonnes in 2024 to 6.4 and 0.7 tonnes, respectively, by 2040. Without implementing a ban on ICEVs (REF scenario), annual NO_x emissions in Leeds declined sharply. In comparison to the REF scenario, the proposed ban on ICEV sales in 2035 (SBP scenario) and 2030 (ABP scenario) caused a decrease of 98 tonnes and 104 tones in annual NO_x emissions by 2040, respectively. Regardless of whether the ban on ICEV sales is implemented or not, the annual emissions of NO_x are expected to decrease gradually in the coming years. Furthermore, the implementation of the ban on ICEV sales could result in the near elimination of NO_x from vehicle exhaust by 2040. Simultaneously, in terms of the overall planning timeframe of 2020–2040, the earlier the implementation of the ban on ICEV sales, the greater the benefit in reducing cumulative NO_x emissions. It is worth mentioning that the current results only took into account NO_x emitted from vehicle pipelines and did not consider the NO_x produced

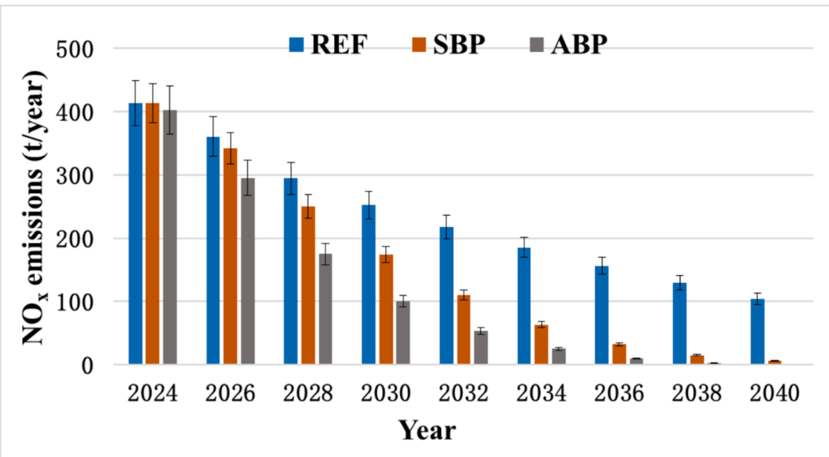

Fig. 5. Annual NO_x emissions under the REF, SBP, and ABP scenarios (The error bars indicate standard deviation).

from electricity production. Moreover, implementing an earlier ban on ICEV sales may have economic and social implications, potentially affecting industries and workers involved in the production, distribution, and maintenance of these ICEVs (Letmathe and Suares, 2020; Plötz et al., 2019). Thus, a prepared implementation of a ban on ICEV sales is recommended to mitigate potential negative impacts on the industry and consumers.

To further ascertain the contribution of different vehicle types to total NO_x emissions, the percentage of NO_x emissions from passenger cars powered by different energy types was calculated under the REF, SBP, and ABP scenarios, as illustrated in Fig. 6. It can be observed that NO_x was predominantly generated by petrol and diesel passenger cars, accounting for 95–99 % of the total emissions across all scenarios. By contrast, PHEVs contributed relatively less under three scenarios, while their contribution proportion was predicted to gradually increase over time. It is worth noting that the present study estimated NO_x emissions for all diesel vehicles based on the NO_x emission limits with varying Euro emission standards. However, previous experimental research demonstrated that under realistic driving conditions, NO_x emissions generated by diesel vehicles significantly exceeded NO_x emission limits (Luján et al., 2018; Weiss et al., 2012). Therefore, it is likely that diesel vehicles are probably responsible for a greater proportion of local NO_x emissions in reality. Compared to the SBP and REF scenarios, the current results under the ABP scenario revealed the contribution of PHEVs to annual NO_x emissions was relatively smaller in 2035 and 2040.

3.2. *Exhaust and non-exhaust $PM_{2.5}$ emissions*

We evaluated the yearly $PM_{2.5}$ emissions in Leeds, taking into account both exhaust and non-exhaust sources, across the REF, SBP, and ABP scenarios, and the results quantified are illustrated in Fig. 7. Unlike NO_x emissions, the total $PM_{2.5}$ emissions showed a consistent upward trend across all scenarios over time. From Fig. 8, ICEVs were the primary source to total $PM_{2.5}$ emissions under the REF, SBP, and ABP scenarios before 2030, contributing up to 92 %. However, BEVs would become the most significant contributor after 2025 in SBP and ABP scenarios, where BEVs accounted for 98 % and almost 100 % of $PM_{2.5}$ emissions in 2040, respectively. Regardless of any one of the three scenarios considered, the relative share of $PM_{2.5}$ emissions from ICEVs (both diesel and petrol cars) to total emissions declined over time. In contrast, the contribution of $PM_{2.5}$ emissions from BEVs increased rapidly under the REF, SBP, and ABP scenarios. In addition, the results in Fig. 7 showed that a ban on ICEV sales under the SBP and ABP scenarios resulted in a marginal rise in total $PM_{2.5}$ emissions in 2040, by 4.5 % and 4.7 %, respectively, compared to that under the REF scenario. This observation is likely ascribed to the fact that implementing a ban on ICEVs (ABP and SBP scenarios) would cause an increase in the market share of BEVs and thus generate a higher amount of non-exhaust $PM_{2.5}$ emissions because of their greater weight relative to other types of vehicles. This implies that the growing adoption of BEVs may not trigger a substantial reduction in $PM_{2.5}$ levels within the confines of prevailing emission regulations. Similarly, Beddows and Harrison (2021) reported that when regenerative braking is not considered, non-exhaust $PM_{2.5}$ emissions generated by BEVs were more than total exhaust and non-exhaust $PM_{2.5}$ emissions generated by equivalent ICEVs. Timmers and Achten (2016) revealed that the widespread adoption of BEVs was unlikely to dramatically reduce PM emissions from the transport sector. This is because non-exhaust sources from passenger cars account for about 85 % of $PM_{2.5}$ emissions, and this proportion is expected to increase in the future.

To further elucidate the causes behind the increase in total $PM_{2.5}$ emissions after the ban on ICEV sales, the individual contributions of various sources to $PM_{2.5}$ emissions were evaluated under three scenarios, and the obtained results are illustrated in Fig. 9. From Fig. 9, regardless of the scenario considered, exhaust $PM_{2.5}$ emissions showed a gradual decline over time, while non-exhaust $PM_{2.5}$ emissions exhibited an upward trend. More specifically, when compared to 2024 levels, annual exhaust $PM_{2.5}$ emissions declined by

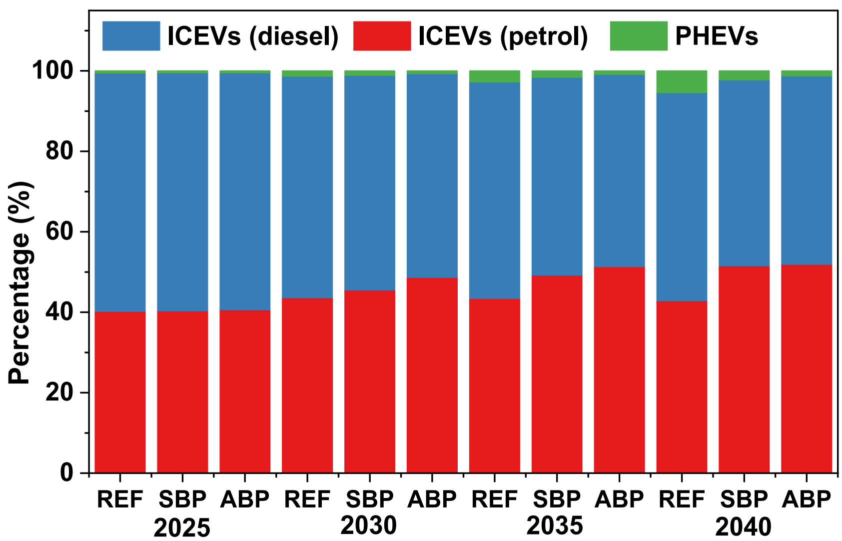

Fig. 6. Percentage of NO_x emissions from different types of passenger cars.

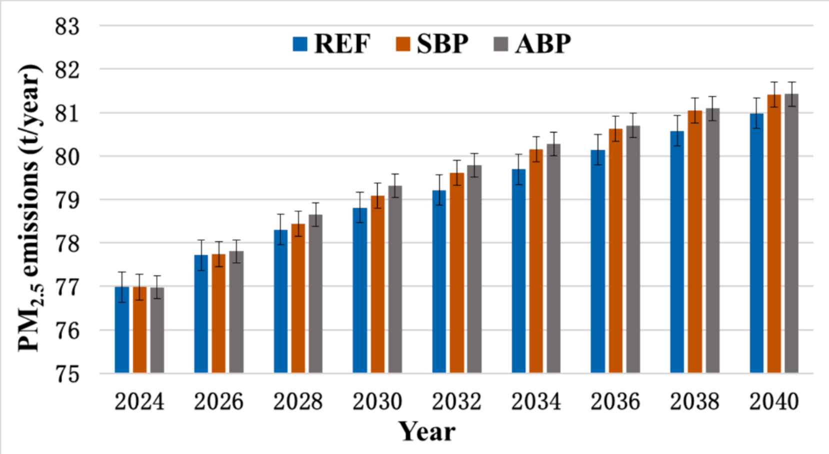

Fig. 7. Annual $PM_{2.5}$ emissions under the REF, SBP, and ABP scenarios (The error bars indicate standard deviation).

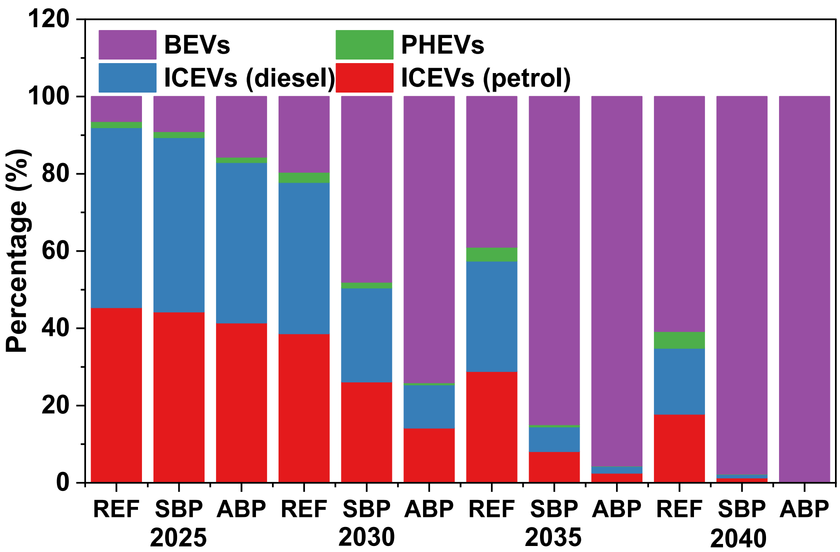

Fig. 8. Proportion of total $PM_{2.5}$ emissions from different types of passenger cars.

63 %, 98 %, and 100 % under the REF, SBP, and ABP scenarios, respectively, by 2040. In contrast, corresponding non-exhaust $PM_{2.5}$ emissions increased by 14 %, 19 %, and 19 %, respectively, by 2040. The opposite trends observed in the changes between the exhaust and non-exhaust $PM_{2.5}$ emissions can be attributed to the following factors. First, the level of exhaust $PM_{2.5}$ was reduced significantly as tailpipe emission standards for ICEVs became increasingly stringent (Chen and He, 2014; Shindell et al., 2011). Second, the REF, SBP, and ABP scenarios were projected to increase the number of BEVs, which emitted more non-exhaust $PM_{2.5}$ emissions owing to their heavier weight relative to ICEVs, as evidenced in Fig. 8 (Hooftman et al., 2018; Lewis et al., 2019; Timmers and Achten, 2016). Previous studies have shown a significant correlation between non-exhaust PM emissions resulting from brake wear, tyre wear, and road wear and vehicle weight (Liu et al., 2021; Liu et al., 2022a; Timmers and Achten, 2016; Timmers and Achten, 2018). The frictional interaction between brake pads and brake discs generates brake wear emissions. The energy dissipated due to friction to slow down vehicle speed is directly proportional to the vehicle's speed and weight (Rajamani et al., 2010). Consequently, the heavier the vehicle weight, the more energy is required to decelerate vehicle speed, which causes an increase in brake wear emissions. In parallel, the frictional interaction between the road surface and the vehicle tyre tread would cause tyre wear and road abrasion. This friction is mainly influenced by the normal force exerted by vehicle tyres on road surfaces and the coefficient of friction between them (Grosch, 2008; Rajamani et al., 2010). The normal force applied by the tyre to the road is positively correlated with vehicle weight. Hence, the increased vehicle weight enhances both abrasion rates of road surface and tyre tread, resulting in elevated tyre and road wear emissions. Simons (2016) calculated the EFs of non-exhaust $PM_{2.5}$ and found that the $PM_{2.5}$ EFs were highest for a large car, followed by a medium car, and then a small car. Wang et al. (2017) explored tyre wear characteristics using a modified theoretical model and identified a direct correlation between increasing vertical load and tyre wear. It means that the increased vehicle weight would lead to

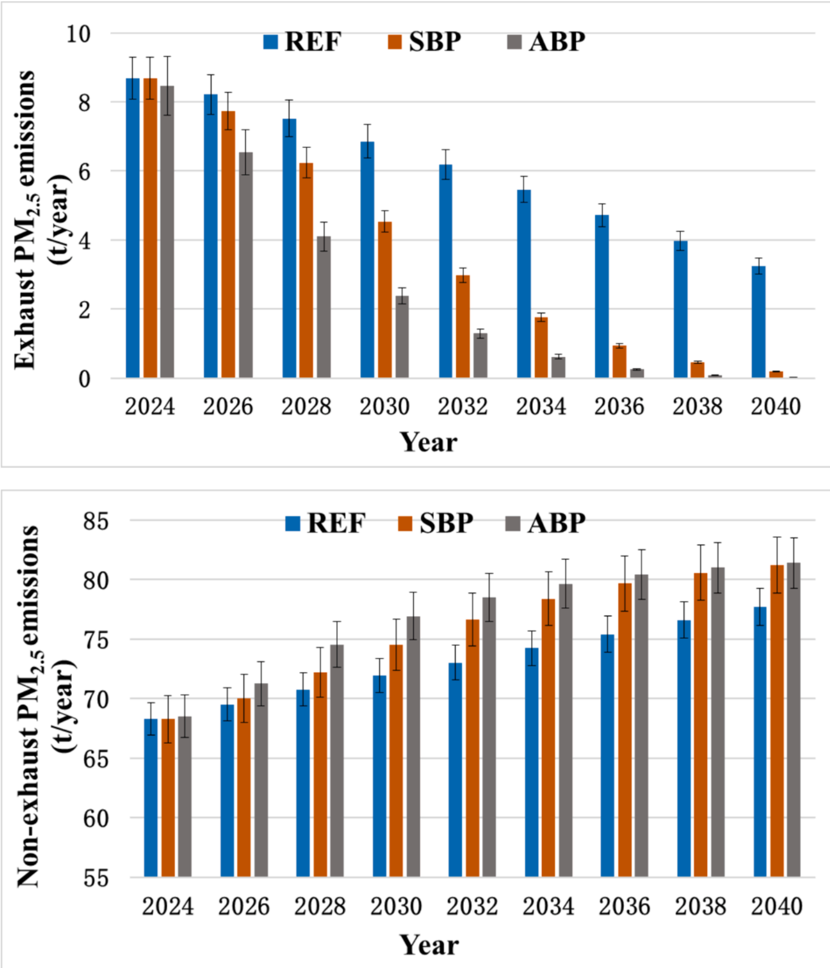

Fig. 9. $PM_{2.5}$ emissions from exhaust and non-exhaust sources under the REF, SBP, and ABP scenarios (The error bars indicate standard deviation).

a proportional rise in tyre wear emissions. Garg et al. (2000) evaluated brake wear emissions from various sizes of passenger cars. The results demonstrated that the brake wear emissions from a larger car were more than 55 % higher than those from a small car.

In comparison to the REF scenario, the annual $PM_{2.5}$ emissions from exhaust sources in 2040 under the SBP and ABP scenarios reduced by 94 % and 99 %, respectively, while non-exhaust $PM_{2.5}$ emissions showed increases of 4.5 % and 4.7 % annually. Close inspection of Fig. 9 discovered that yearly $PM_{2.5}$ emissions from non-exhaust sources significantly exceeded those from the exhaust source. Specifically, exhaust sources contributed less than 4 % to total $PM_{2.5}$ emissions in 2040 in either the REF, ABP or SBP scenarios. This suggests that efforts aimed solely at decreasing vehicle exhaust $PM_{2.5}$ may not be sufficient in effectively reducing $PM_{2.5}$ levels (Kousoulidou et al., 2008). Thus, it is required to introduce new policies specifically targeting non-exhaust $PM_{2.5}$ emissions. The recent proposal for Euro 7 emission standards introduced, for the first time, limits on PM emissions emitted from tyres and brakes (European Environment Agency, 2022), representing a significant step towards addressing non-exhaust PM emissions. To accelerate the mitigation of non-exhaust PM emissions, it is recommended to implement the following enhancements: (1) The government should introduce incentives to promote research and development of advanced technologies geared towards lightweight vehicles, as well as the creation of abrasion-resistant materials for use in tyres and brakes (Gustafsson, 2018; Kupiainen and Pirjola, 2011; Son and Choi, 2022; Timmers and Achten, 2016). (2) As many passenger cars as possible are equipped with regenerative braking systems, as this technology has proven to be highly effective in reducing brake wear emissions, especially on urban roads (Bondorf et al., 2023; Gramstat, 2018). (3) Regular vehicle maintenance and proper tyre inflation are important measures to reduce non-exhaust PM emissions (Fussell et al., 2022; Lewis et al., 2019; Pohrt, 2019). (4) Eco-friendly driving practices are beneficial to reducing vehicle exhaust and non-exhaust emissions (David et al., 2018; Kwak et al., 2013; Liu et al., 2022b; UK, 2023).

For policymakers, it is essential to identify the most significant sources of non-exhaust $PM_{2.5}$ emissions to develop targeted mitigation strategies. The yearly $PM_{2.5}$ emissions of the primary non-exhaust sources were determined, including tyre, brake, and road wear, from various passenger car types under three scenarios, as depicted in Fig. 10. It is evident that the most contributor to $PM_{2.5}$ emissions in all three scenarios was tyre wear, followed by road wear, and then brake wear. It is important to mention that accurate data on the potential impact of regenerative braking technology on brake wear $PM_{2.5}$ emissions is not yet available. Consequently, this

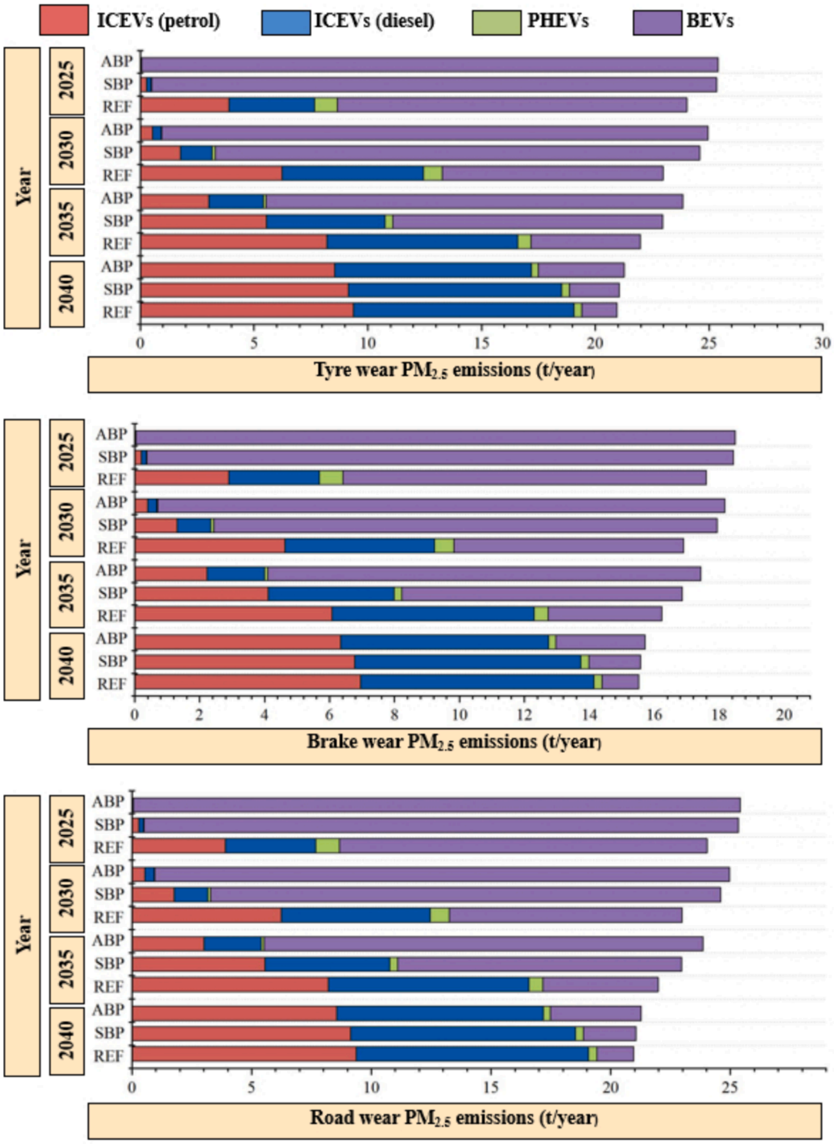

Fig. 10. $PM_{2.5}$ emissions from non-exhaust sources under the REF, SBP, and ABP scenarios.

factor was not considered in our analysis. If regenerative braking technology is taken into account, the contribution of brake wear on $PM_{2.5}$ emissions would be even smaller. In terms of passenger car types, $PM_{2.5}$ emissions from brake, tyre, and road wear were reduced for conventional passenger cars. However, the corresponding non-exhaust emissions from BEVs were expected to increase due to their growing market share and higher vehicle weight. After implementing the ban on ICEV sales under the ABP scenario, the $PM_{2.5}$ emissions from brake, tyre, and road wear in BEVs were found to be noticeably higher than those in other types of passenger cars, and even exceeded overall emissions produced by other types of passenger cars.

3.3. Analysis of health risks

In this study, the EHI associated with $PM_{2.5}$ and NO_x emissions was examined to assess health effects across three scenarios of the REF, SBP, and ABP. The combined EHI results linked to $PM_{2.5}$ and NO_x are depicted in Fig. 11. From Fig. 11, there was a gradual decrease in combined EHI values attributed to $PM_{2.5}$ and NO_x emissions over time across all REF, SBP, and ABP scenarios. Among these scenarios, the EHI values under the ABP scenario were noticeably lower than those under the REF and SBP scenarios for 2026–2032. After 2034, the gap between the EHI values under the REF and SBP scenarios progressively narrowed. It means that the earlier the ban on ICEV sales is implemented, the greater the potential cumulative health benefits will be. In addition, when the time horizon under consideration is stretched, the impact of banning ICEV sales becomes gradually weaker over time. Therefore, the difference in EHI

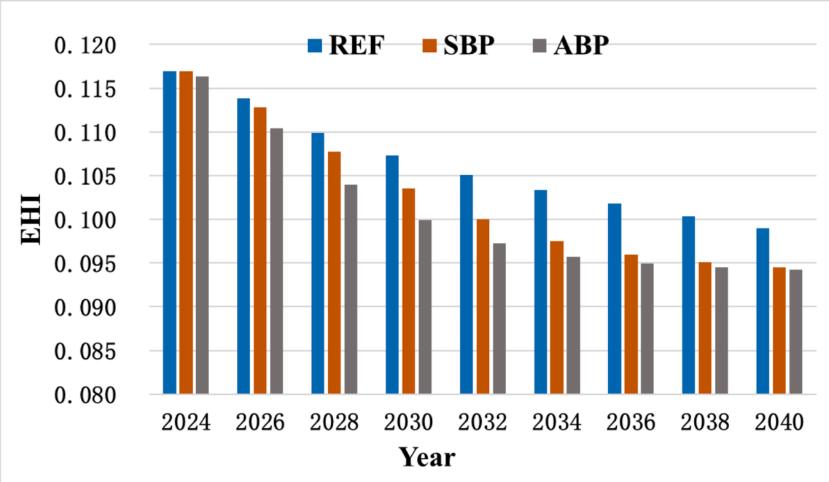

Fig. 11. Combined EHI values associated with NO_x and $PM_{2.5}$.

values between the ABP and SBP scenarios would also be minor once the policy of banning ICEV sales has increased the market share of BEVs to a certain level.

To determine the individual contributions of $PM_{2.5}$ and NO_x emissions to health effects, the individual EHI values related to $PM_{2.5}$ and NO_x were computed independently, and the outcomes are illustrated in Fig. 12. The EHI values associated with NO_x emissions exhibited a sharp reduction over time across the three scenarios studied. The EHI values under the ABP and SBP scenarios were lower compared to those observed under the REF scenario. Although the EHI values for SBP and ABP scenarios were similar in 2040, it is obvious that the cumulative EHI associated with NO_x was higher for the SBP scenario than for the ABP scenario. This finding indicates that the implementation of an earlier ban on ICEV sales could contribute to greater health benefits. Compared to the REF and SBP scenarios, the magnitude of the rise in cumulative EHI values associated with $PM_{2.5}$ emissions was more pronounced in the ABP scenario over time. This implies that the ban on ICEV sales may not yield any health benefits, given the current levels of $PM_{2.5}$ emissions from non-exhaust sources of both BEVs and ICEVs. Therefore, to substantially lower the levels of $PM_{2.5}$ emissions and obtain significant health benefits by banning ICEV sales, it is required to introduce stringer regulations on non-exhaust $PM_{2.5}$ emissions.

The annual number of premature deaths associated with $PM_{2.5}$ and NO_x emissions was calculated under the REF, SBP, and ABP

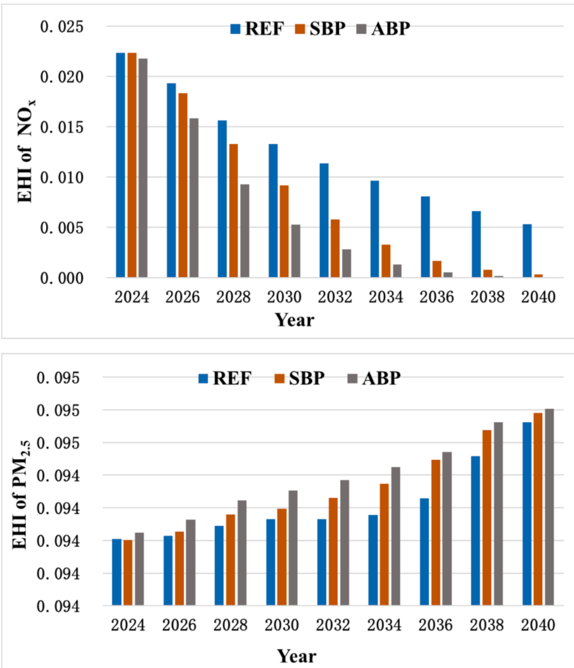

Fig. 12. Individual EHI values for NO_x and $PM_{2.5}$.

scenarios, and the results calculated are listed in Table 2. Regarding the REF scenario, the annual number of premature deaths attributed to NO_x emissions from passenger cars in Leeds showed a reduction from 37 in 2024 to 10 in 2040, whereas the corresponding death toll associated with $PM_{2.5}$ emissions increased from 20 in 2024 to 28 in 2040. Under both the SBP and ABP scenarios, analogous findings were observed concerning the annual death toll associated with $PM_{2.5}$ and NO_x emissions. From Table 2, annual deaths associated with combined $PM_{2.5}$ and NO_x emissions have been falling over time under the REF, SBP, and ABP scenarios. In 2040, it was projected that the annual total number of deaths due to $PM_{2.5}$ and NO_x from passenger cars in Leeds will be 38 (95 % CI: 25, 51), 30 (95 % CI: 21, 40), and 29 (95 % CI: 21, 39) under the REF, SBP, and ABP scenarios, respectively. It is worth mentioning that the emissions of precursors, such as sulfur dioxide (SO_2) and NO_x, significantly contribute to the formation of SIA $PM_{2.5}$, including sulfate (SO_4^{2-}) and nitrate (NO_3^-) (Mehlig et al., 2021; Hooftman et al., 2016). Consequently, the atmospheric concentrations of $PM_{2.5}$ are likely underestimated without considering the formation of SIA $PM_{2.5}$ resulting from NO_x in the current work. Close inspection of Table 2 showed that, compared to the death toll in 2022, changes in $PM_{2.5}$ and NO_x emissions under the REF, SBP, and ABP scenarios would avoid 19, 27, and 28 deaths in Leeds by 2040, respectively. Nevertheless, it is notable that the population in Leeds only account for 1.16 % of the total population of the UK. Assuming that the concentrations of $PM_{2.5}$ and NO_x in the UK are the similar to those with the current work, a ban on ICEV sales in either the ABP scenario or the SBP scenario, compared to the REF scenario, could avoid 690 or 776 premature deaths between 2024 and 2040 in the UK. These results revealed that the implementation of an earlier ban on ICEV sales contributes to greater health benefits, mainly due to the substantial reduction in NO_x emissions.

4. Key parameter analysis

4.1. Non-exhaust emission factors

Non-exhaust EFs from BEVs, including brake, tyre, and road wear, were determined using the EMEP/EEA Guideline approach, with adjustments made for the impact of vehicle weight (Beddows and Harrison, 2021; Harrison et al., 2021; Liu et al., 2021; EU, 2019). Given the nascent stage of non-exhaust emission studies for BEVs and ICEVs, it remains a challenge to determine completely accurate and definitive non-exhaust EFs. In this context, a further analysis was performed to ascertain the effects of non-exhaust emission factors on annual $PM_{2.5}$ emissions and corresponding human health. This analysis employed three different values for the non-exhaust EFs: C_1, representing a 15 % decrease in EFs for all vehicle classes; C_2, assuming that EFs of ICEVs remain constant, while EFs of BEVs and PHEVs are equivalent to that of ICEVs to simulate the rapid development of future BEV technology; and C_3, reflecting a 15 % increase in EFs for all vehicle classes. The results derived from this analysis are illustrated in Fig. 13, where the labels R, S, and A correspond to the REF, SBP, and ABP scenarios, respectively. As expected, the $PM_{2.5}$ emissions across all scenarios exhibited a substantial reduction as non-exhaust EFs decrease over a range of + 15 % to −15 %. Correspondingly, the associated death toll decreased by 24 %, 25 %, and 25 % for the REF, SBP, and ABP scenarios, respectively. In addition, when the emission factor of BEVs falls to the same level as that of ICEVs, the ban on ICEV sales can reduce $PM_{2.5}$ by 3.5 % and total deaths by 4.2 % in 2040. Therefore, it is necessary to undertake effective measures aimed at reducing non-exhaust emissions especially those from BEVs, which, in turn, would have a beneficial effect on human health.

Table 2
Annual number of premature deaths (95% CI) attributed to $PM_{2.5}$ and NO_x emissions under the REF, SBP, and ABP scenarios.

Scenario	Pollutants	2024	2028	2032	2036	2040
REF	$PM_{2.5}$ and NO_x	57 (34, 78)	48 (30,66)	45 (28,60)	41 (26,55)	38 (25,51)
	$PM_{2.5}$	20 (13, 26)	22 (15,29)	25 (17,32)	27 (18,35)	28 (19,37)
	NO_x	37 (21, 52)	26 (15,37)	20 (11,28)	14 (8,20)	10 (6,14)
SBP	$PM_{2.5}$ and NO_x	57 (34, 78)	45 (29,62)	36 (24,48)	31 (21,42)	30 (21,40)
	$PM_{2.5}$	20 (13, 26)	23 (16,30)	26 (18,34)	28 (19,37)	29 (20,39)
	NO_x	37 (21, 52)	22 (15,32)	10 (6,14)	3 (2,5)	1 (0,1)
ABP	$PM_{2.5}$ and NO_x	57 (34, 77)	39 (25,53)	31 (21,42)	29 (20,40)	29 (21,39)
	$PM_{2.5}$	20 (13, 26)	23 (16,31)	26 (18,35)	28 (19,38)	29 (21,39)
	NO_x	36 (21, 51)	16 (9,22)	5 (3,7)	1 (0,2)	0 (0,0)

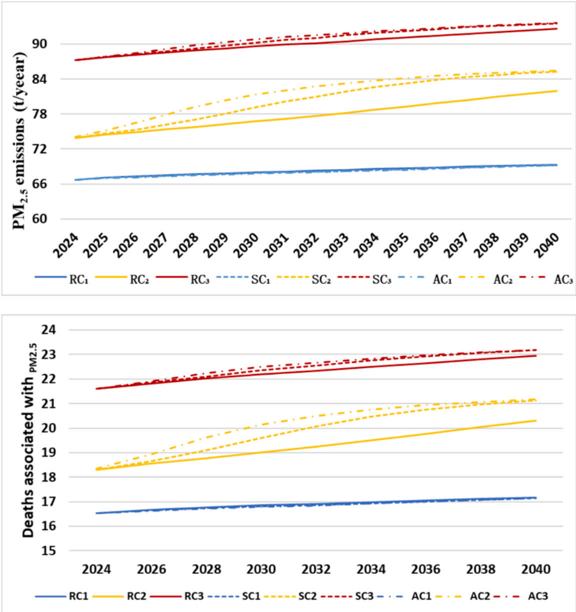

Fig. 13. $PM_{2.5}$ emissions and associated death toll with the change in the non-exhaust emission factor.

4.2. Mileage-age rates

The mileage-age rate is a critical factor in determining the annual mileage for vehicles of varying ages, which, in turn, affects the vehicle's annual emissions. To date, the mileage-age rate for BEVs has not been conclusively determined due to their relatively recent introduction into the market. Only a recent qualitative study by Mehlig et al. (2023) disclosed that with increasing vehicle age, the annual mileage of BEVs decreases faster than that of ICEVs. In this research, a sensitivity analysis was performed to explore the impact on $PM_{2.5}$ emissions and human health, employing three progressively decreasing mileage-age rates for BEVs: the average rate of diesel and petrol vehicles (denotes as C_1), the rate for petrol vehicles (C_2), and 95 % of the petrol vehicle rate (C_3). The obtained results are shown in Fig. 14, where R, S, and A represent the REF, SBP, and ABP scenarios, respectively. From Fig. 14, regardless of the REF, SBP, and ABP scenarios, there was a notable reduction in annual $PM_{2.5}$ emissions and associated death toll with a decrease in the mileage-age rate from C_1 to C_3. Among them, the counterparts for the ABP scenario showed the largest decrease. More specifically, when comparing the case of AC_3 to AC_1 in 2040, annual $PM_{2.5}$ emissions and associated death toll were reduced by 22 % and 24 %, respectively. It means that earlier implementation of the ban on ICEV sales is more effective in reducing $PM_{2.5}$ emissions and more beneficial to human health as the mileage-age rate of BEVs reduces. Therefore, to maximise environmental and human health benefits, it is necessary to reduce BEV mileage by encouraging people to travel by carpooling or public transport, in conjunction with the earlier implementation of the ban on ICEV sales.

4.3. Vehicle weight

Vehicle weight is a crucial factor influencing non-exhaust EFs, which subsequently impact the environment and human health. In this study, a sensitivity analysis was conducted to assess the effects of vehicle weight on both $PM_{2.5}$ emissions and the number of premature deaths. In the current work, C1 refers to a 5 % increase in all vehicle weight; C2 represents 15 % decrease in vehicle weight; and C3 corresponds to a 30 % reduction in vehicle weight. The labels R, S, and A represent the REF, SBP, and ABP scenarios, respectively. The outcomes obtained from the scenarios above are shown in Fig. 15. It can be observed that the annual $PM_{2.5}$ emissions and the number of premature deaths decreased over time when vehicle weight was reduced. More specifically, the number of premature deaths was reduced by up to 15 % under the ABP scenario when vehicle weight was decreased by 30 %. Compared to other scenarios, the annual $PM_{2.5}$ emissions and the number of premature deaths under the ABP scenario showed a more pronounced reduction. It means that implementing the policy of banning ICEVs early, along with simultaneously reducing vehicle weight, would be more effective in reducing $PM_{2.5}$ emissions and bringing greater benefits to human health.

5. Conclusion

In the current work, the annual $PM_{2.5}$ and NO_x emissions in Leeds, UK, along with the corresponding health effects were evaluated using an integrated model called 'Fleet-Emission-Health' when introducing the ban on the sales of internal combustion engine vehicles (ICEVs) within different timeframes. The results showed that when not considering emissions generated from electricity generation,

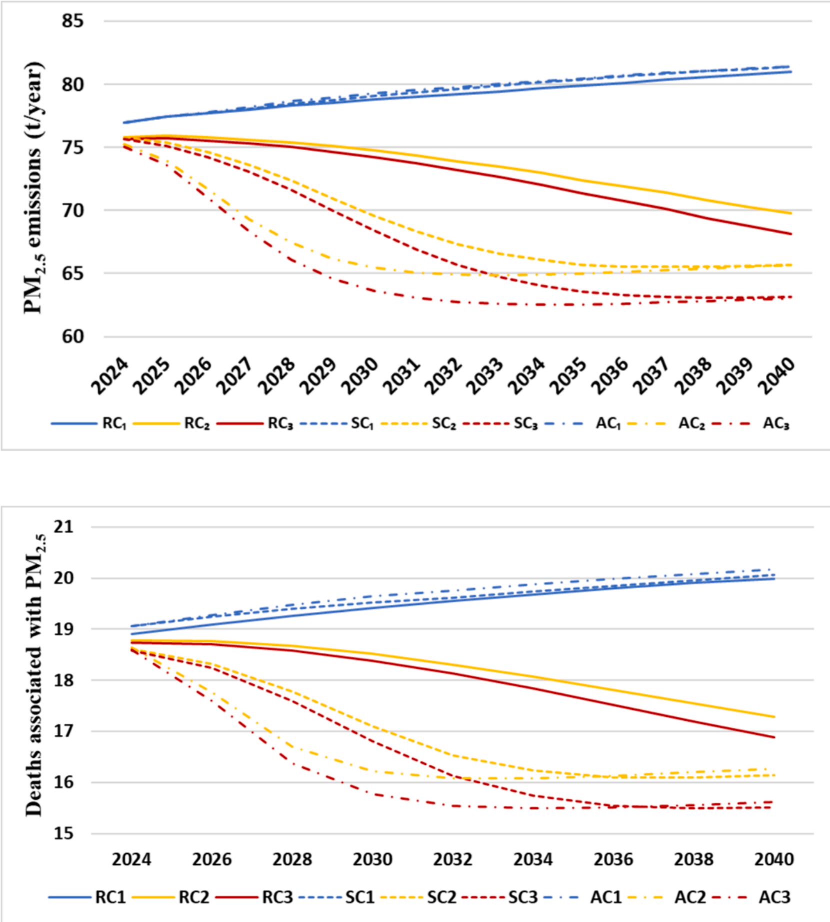

Fig. 14. $PM_{2.5}$ emissions and associated death toll with the change in the BEV mileage-age rates.

banning ICEV sales would trigger a maximum reduction up to almost 100 % in annual NO_x emissions by 2040, whereas overall $PM_{2.5}$ emissions would be slightly increased owing to higher non-exhaust $PM_{2.5}$ emissions emitted from battery electric vehicles (BEVs). In terms of health effects, the combined emission harm equivalent index (EHI) associated with NO_x and $PM_{2.5}$ emissions was reduced by up to 19 %. More specifically, implementing a ban on ICEV sales in 2030 would avoid 28 deaths associated with NO_x and $PM_{2.5}$ emissions in Leeds by 2040 compared to those in 2022. Moreover, the results derived from the sensitivity analysis revealed that the non-exhaust emission factors, mileage-age rate, and vehicle weight are crucial determinants influencing the level of emissions and their consequential implications for human health. These findings demonstrate that, to attain greater environmental and health benefits, there is a need to introduce more stringent legislative measures on vehicle non-exhaust emissions and develop a well-reason and effective travel planning.

CRediT authorship contribution statement

Ye Liu: Writing – original draft, Methodology, Investigation, Data curation, Conceptualization. **Haibo Chen:** Writing – review & editing, Validation, Supervision. **Like Jiang:** Writing – review & editing. **Tiezhu Li:** Writing – review & editing, Conceptualization. **Junhua Guo:** Writing – review & editing. **Tangjian Wei:** Writing – review & editing, Software. **Richard Crowther:** Software, Resources.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

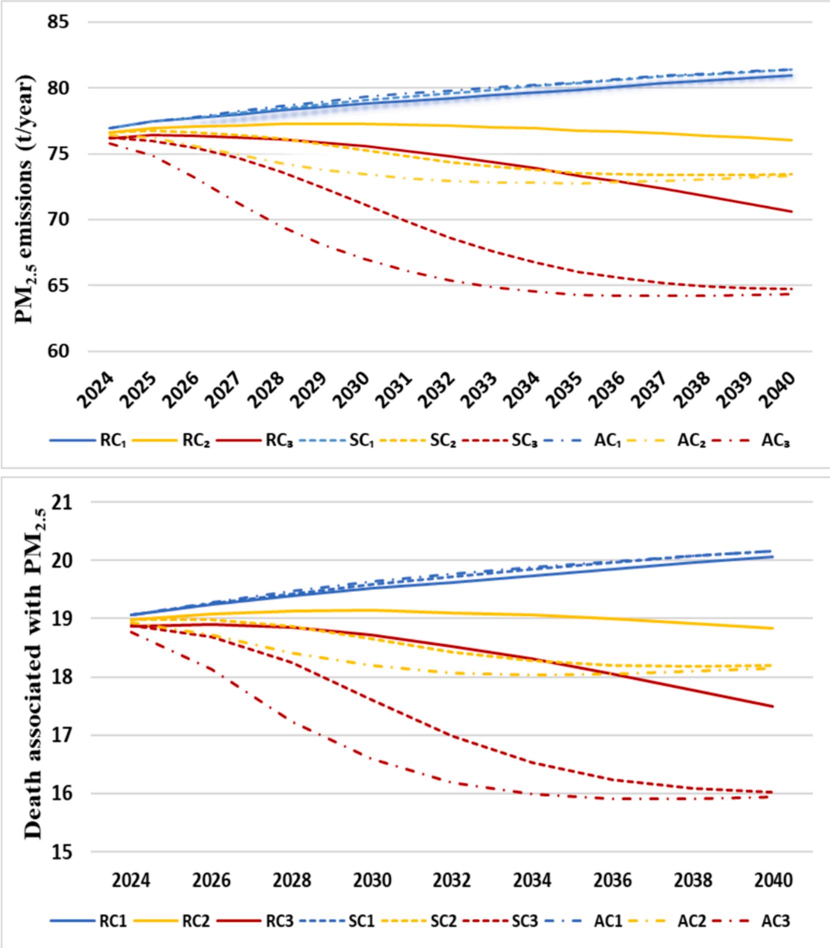

Fig. 15. $PM_{2.5}$ emissions and associated death toll with the change in vehicle weight.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.trd.2024.104343.

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