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Impact of change in traffic flow on vehicle non-exhaust $PM_{2.5}$ and PM_{10} emissions: A case study of the M25 motorway, UK

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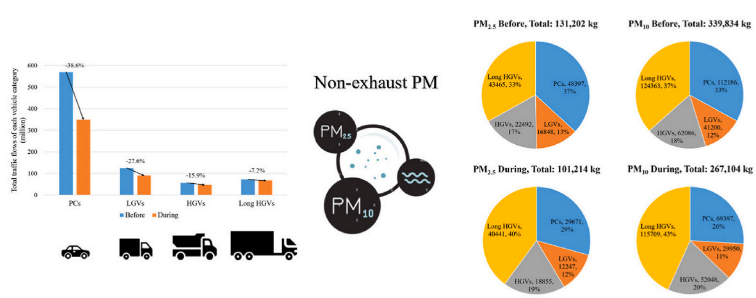
HIGHLIGHTS

- Traffic flow was quantified on the M25 before and during the year of the outbreak.
- Total non-exhaust $PM_{2.5}$ and PM_{10} mass of four categories of vehicles were evaluated.
- Long HGVs emitted the most non-exhaust particles, followed by HGVs, LGVs and PCs.
- Resuspension of road dust was the largest contributor to non-exhaust particles.

GRAPHICAL ABSTRACT



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ABSTRACT

This study quantifies the change in traffic flow on the M25 motorway in the UK due to the COVID-19 outbreak. Moreover, the impact of the change in traffic flow on non-exhaust $PM_{2.5}$ and PM_{10} emissions for different categories of vehicle was explored. During the year of the COVID-19 outbreak (March 2020 to February 2021), the total traffic flows of passenger cars (PCs), light goods vehicles (LGVs), heavy goods vehicles (HGVs), and long HGVs on the M25 motorway decreased by 38.6%, 27.6%, 15.9% and 7.2%, respectively, in comparison to the previous year. Correspondingly, the total mass of non-exhaust emissions ($PM_{2.5}$ and PM_{10}) of PCs, LGVs, HGVs, and long HGVs reduced by 38.7%, 27.3%, 16.2% and 7%, respectively. The traffic flows per year before and during the COVID-19 outbreak of long HGVs were 87.2% and 80.7% less than those of PCs. Correspondingly, the long HGVs emitted 10.2% less but 36.3% more $PM_{2.5}$ emissions, as well as 10.9% and 66.7% more PM_{10} emissions than the latter, indicating that long HGVs contribute much more to non-exhaust particles than PCs. In addition, it was found that resuspension of road dust on the M25 motorway was the largest contributor to air pollution among non-exhaust emissions, followed by road wear, tyre wear, and brake wear particles.

1. Introduction

Recently, more attention has been paid to vehicle non-exhaust particles due to the adverse effects on human health and the environment (Amato et al., 2014; Ostro et al., 2011; Wang et al., 2022; Willers et al., 2013). Exhaust and non-exhaust emissions from road vehicles are the major source of air pollution (García-Contreras et al., 2021; Goel and Kumar, 2014; Pant and Harrison, 2013). The former includes nitrogen

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oxides (NO_x), carbon monoxide (CO), and exhaust particulate matter (PM), and the latter usually refers to tyre wear, brake wear, road wear particles, and resuspension of road dust. To meet increasingly strict emission regulations, exhaust emissions have been effectively reduced due to the rapid development of engine combustion control and after-treatment technologies (Mahesh et al., 2019; Shindell et al., 2011; Wang et al., 2019).

However, vehicle non-exhaust emissions have not been regulated effectively. Previous studies have found that non-exhaust $PM_{2.5}$ and PM_{10} emissions have exceeded exhaust ones as a major part of road traffic emissions. According to the UK National Atmospheric Emissions Inventory (AQEG, 2019), tyre wear, brake wear, and road wear emissions climbed from 26% of road traffic $PM_{2.5}$ emissions in 2000 to 67% in 2018, and PM_{10} increased from 39% in 2000 to 73% in 2018. In the U. S., exhaust emissions were predicted to continue declining until 2035; in contrast, there has been a growing trend in non-exhaust emissions. The National Emissions Inventory (U.S. Environmental Protection Agency, 2019) showed that non-exhaust emissions increased to 98% of total major PM_{10} emissions from road traffic by 2020. Regarding $PM_{2.5}$, non-exhaust emissions have accounted for more than 80% of road traffic emissions since 2015, and this share was expected to increase to 97% by 2030. The shares in PM_{10} and $PM_{2.5}$ emissions will remain stable until 2035. Emission inventories for several cities and regions in Latin America also demonstrated the same result of non-exhaust emissions far outstripping exhaust emissions as the principal source of road traffic emissions (Pachón et al., 2018; Secretaria del Medio Ambiente de la Ciudad de Mexico, 2018). In addition, increasing studies have reported the negative consequences of exposure to non-exhaust particles. For instance, among all critical risk factors for mortality, exposure to ambient fine particulate matter is ranked seventh (OECD, 2017). The toxicological evidence demonstrates the adverse health impacts of chemical components in non-exhaust particles (Grigoratos and Martini, 2014), especially on respiratory and cardiovascular morbidity (Ostro et al., 2011; Tonne et al., 2016; Willers et al., 2013). As a result, further research on non-exhaust particles is needed.

Many investigators have examined that the traffic flow change inevitably leads to fluctuations in vehicle emissions. Liu et al. (2011) applied a novel Traffic And Vehicle Emissions Linkage approach to inspect how traffic control limits vehicle emissions. A model developed by Noland and Quddus (2006) showed that initial emissions reduction benefits may potentially be slashed by increased traffic. Xue et al. (2013) established a model to investigate the increase of traffic flow on traffic congestion and air pollution in Beijing. However, non-exhaust emissions are under-explored in the existing literature. In this context, The COVID-19 pandemic is not only a risk for health and economic well-being but also has greatly affected travel over the past two years (Budd and Ison, 2020; Shepherd et al., 2021), contributing to the prediction of the impact of traffic flow change on vehicle emissions. For instance, during the COVID-19 lockdown, there were 53% VOC and 76% NO_x emissions reductions in Beijing's transport sector (Lv et al., 2020). Tanzer-Gruener et al. (2020) found that, during the COVID-related closures in Pittsburgh, USA, the proportional reduction in CO and NO_2 concentrations due to the morning peak was consistent with the proportional reduction in commuter traffic flows at the high traffic locations, approximately 50%. The UK monthly average daily traffic flow in April 2020 was 69% lower than the same month the previous year, and quantification of the change in air quality over this period showed a 38.3% reduction in NO_2 and 16.5% reduction in $PM_{2.5}$ concentrations (Jephcote et al., 2021). However, to the best of our authors' knowledge, current information regarding the effect of change in traffic flow on different categories of vehicle non-exhaust emissions is limited. Therefore, the purpose of this work is to quantify the change in traffic flows of different categories of vehicles on the M25 motorway, a representative UK motorway, before and during one year of the COVID-19 outbreak. In addition, the effect of change in these traffic flows on vehicle non-exhaust particles is explored.

2. Methods

2.1. Data collection and processing

M25, as a typical motorway in the UK, was chosen to conduct this research. This orbital motorway encircles most of Greater London, one of the UK's most important and busiest motorways, connecting to key transport hubs such as Dartford Crossing, Gatwick Airport, and Heathrow Airport (Fig. 1). Existing studies have explored the huge traffic flows, severe congestion and environmental pollution problems of the M25 (Kalair et al., 2021; Sayegh et al., 2018; Wang et al., 2009). Minute traffic flow data of four different categories of vehicles on the M25 motorway from March 2019 to February 2021 were obtained from National Highways MIDAS (Motorway Incident Detection and Automatic Signalling) system. This MIDAS system collected data using inductive loop sites at approximately 400–500 m intervals. The whole M25 motorway contains 30 junctions, and the distance between every two adjacent junctions was measured using Google Directions.

The traffic flows of the middle five consecutive sites between each adjacent junction were calculated. Python code was used to process the data by month and divide it into two time periods divided by the COVID-19 outbreak time. The first period was from March 2019 to February 2020, which refers to the year before the COVID-19 outbreak, and the second was from March 2020 to February 2021, which refers to the first year of the COVID-19 outbreak.

The data of weight and length of vehicles using the road section during either time period are summarised in Table S1 of the Supplementary material according to the SMMT (2022), the UK's primary source of vehicle data on the motor industry, and Hunt's Transport (2022). From these data, four categories of vehicles were classified according to the MIDAS system by identifying the length of each vehicle category, which corresponds mainly to the PCs, LGVs, HGVs, and long HGVs. More detailed information on the average vehicle length and weight of different categories of vehicles is summarised in Table 1. The average gross weight of PCs was calculated by summing the vehicle kerb weight and the average weight of three adults (75 kg each).

2.2. Calculation of non-exhaust emissions

The non-exhaust $PM_{2.5}$ and PM_{10} emissions for the year before and during the COVID-19 outbreak were calculated using the following equations

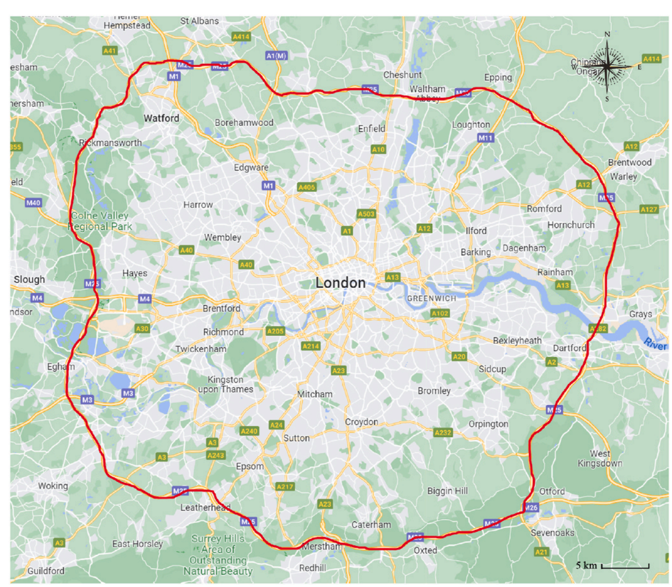

Fig. 1. M25 motorway map.

Table 1
Information regarding the length and weight of various categories of vehicles.

Vehicle Category	Vehicle Length (m)	Main Vehicle	Average Vehicle Length (m)	Average Vehicle Gross Weight (kg)
Category 1	$L < 5.2$	PCs	4.3	1899.8
Category 2	$5.2 \leq L < 6.6$	LGVs	5.5	3653.3
Category 3	$6.6 \leq L < 11.6$	HGVs	10.0	17277.8
Category 4	$L \geq 11.6$	Long HGVs	12.6	28571.4

$$M_k = \sum_{j=1}^{4} \sum_{i=1}^{29} EF_{jk} \times D_i \times N_{ij} \quad (1)$$

$$M = \sum_{k=1}^{4} M_k \quad (2)$$

where M_K *is* the mass of various types of non-exhaust emissions, M is the total non-exhaust emission mass, D_i means the distance between the two adjacent junctions that defined road segment i, N_{ij} is the traffic flow of vehicle category j on road segment i, and EF_{jk} is the emission factor of non-exhaust $PM_{2.5}$ and PM_{10} emissions by vehicle category j and non-exhaust emission category k, which was obtained based on the model developed by Beddows and Harrison (2021). There are four steps in the model building: 1) the non-exhaust $PM_{2.5}$ and PM_{10} EFs for different vehicle categories on motorway roads in national inventories were adopted (Lewis et al., 2019); 2) the different categories of vehicle mass were estimated; 3) the correlations between tyre wear, brake wear, road wear and road dust resuspension EF and vehicle weight were determined by fitting the data using the non-linear least-squares method. The equation is defined as

$$EF = b \cdot W_{ref}^{\frac{1}{c}} \quad (3)$$

where W_{ref} is vehicle mass divided by 1000 kg; b (mg km^{-1} veh^{-1}) and c (no unit) are regression coefficients whose estimated values are listed in Table 2. The minimum, mean, and maximum values of non-exhaust *EF* were determined according to the mean and extreme values of parameters b and c, thus calculating the minimum, mean, and maximum values of total non-exhaust particles. The minimum and maximum values were marked in the results by means of error bars.

3. Results and discussion

3.1. Traffic flow change

The total traffic flows of four vehicle categories per year before and during the outbreak were compared, and the results are shown in Fig. 2. Compared with the year before the outbreak, total traffic flows during the outbreak of the four vehicle categories decreased by 38.6%, 27.6%, 15.9%, and 7.2%, respectively. A similar finding was reported by

Table 2
Regression coefficient used (b and c) to fit the emission factors vs vehicle weight.

		Motorway	
		b	c
Tyre wear	$PM_{2.5}$	3.8 ± 0.3	2.3 ± 0.4
	PM_{10}	5.5 ± 0.4	2.3 ± 0.4
Brake wear	$PM_{2.5}$	0.4 ± 0.4	1.3 ± 0.4
	PM_{10}	1.0 ± 1.0	1.3 ± 0.4
Road wear	$PM_{2.5}$	2.8 ± 0.5	1.5 ± 0.1
	PM_{10}	5.1 ± 0.9	1.5 ± 0.1
Resuspension of road dust	$PM_{2.5}$	2.0 ± 0.8	1.1 ± 0.4
	PM_{10}	8.2 ± 3.2	1.1 ± 0.4

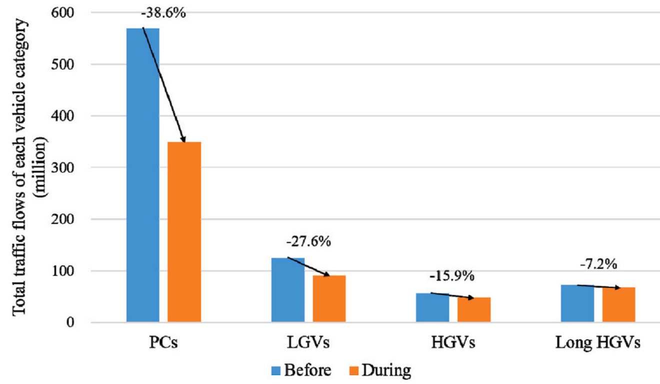

Fig. 2. Total traffic flows of each vehicle category before and during the outbreak.

Hadjidemetriou et al. (2020); they found that human mobility in the UK overall reduced by around 20% after the lockdown was imposed. Specifically, it can be observed that the COVID-19 outbreak severely affected the traffic flow of PCs, while the LGVs and HGVs were less affected. Panzone et al. (2021) also pointed out that more LGVs and HGVs were used for delivery due to the increasing number of orders online, which may explain why these two types of vehicles were less affected. This view has been evidenced by Carrington (2020).

Fig. 3 illustrates the monthly traffic flows of four categories of vehicles before and during the outbreak. Prior to the outbreak, traffic flows of the four vehicle categories did not vary significantly. Although some studies have suggested that more frequent rain and snow conditions during autumn and winter in the UK may negatively affect travel behaviour and traffic flow (Hooper et al., 2014; Zanni and Ryley, 2015), our current results did not illustrate this clearly. This is likely because the M25 motorway connects many storage centres and transport hubs, especially Heathrow Airport, which is highly busy throughout the year.

In contrast, traffic flows varied considerably during the year of the outbreak. Policy restrictions are likely to be the chief reason for the variation in traffic flow in this period (Anzai et al., 2020; Nouvellet et al., 2021; Shepherd et al., 2021; Spelta and Pagnottoni, 2021). The UK government imposed three national lockdowns to control the virus transmission and protect public health capacity. The first lockdown was from 23 March to June 23, 2020, the second from 31 October to December 2, 2020, and the third from 6 January to March 8, 2021. From Fig. 3, there was a substantial drop during the first lockdown for all vehicle categories. Especially in April, traffic flows of each vehicle category reached the lowest number. However, a quick rebound in traffic flows of the four vehicle categories was observed from May, which is likely closely related to the following factors: (1) a relaxing of the restriction rules was introduced, where people were permitted to leave home for activities with a restricted number of people (from May), as well as allowing most hospitality businesses to reopen (from July); (2) individual perceptions and behaviours varying in response to changes in the severity, policy and infrastructure implementations of COVID-19 (Ozbilen et al., 2021), in particular the public's perceived risk reduction due to the relaxation of lockdown restrictions leading to an increased travel willingness related to a rapid rebound in traffic.

In the period of the second national lockdown in November 2020, traffic flows of all vehicle categories again declined steadily, with a noticeable drop in PCs. No rapid rebound occurred in the same way as had happened after the lockdown was lifted for the first time, except for PCs. Such a different trend between PCs and the other three categories of vehicles is likely associated with the increased demand for family trips due to the Christmas holidays. The traffic flows for four categories of vehicles were reduced on the M25 motorway due to the third lockdown in January 2021. There was the largest decline of the proportion for PCs compared with the other three vehicle types. However, as expected,

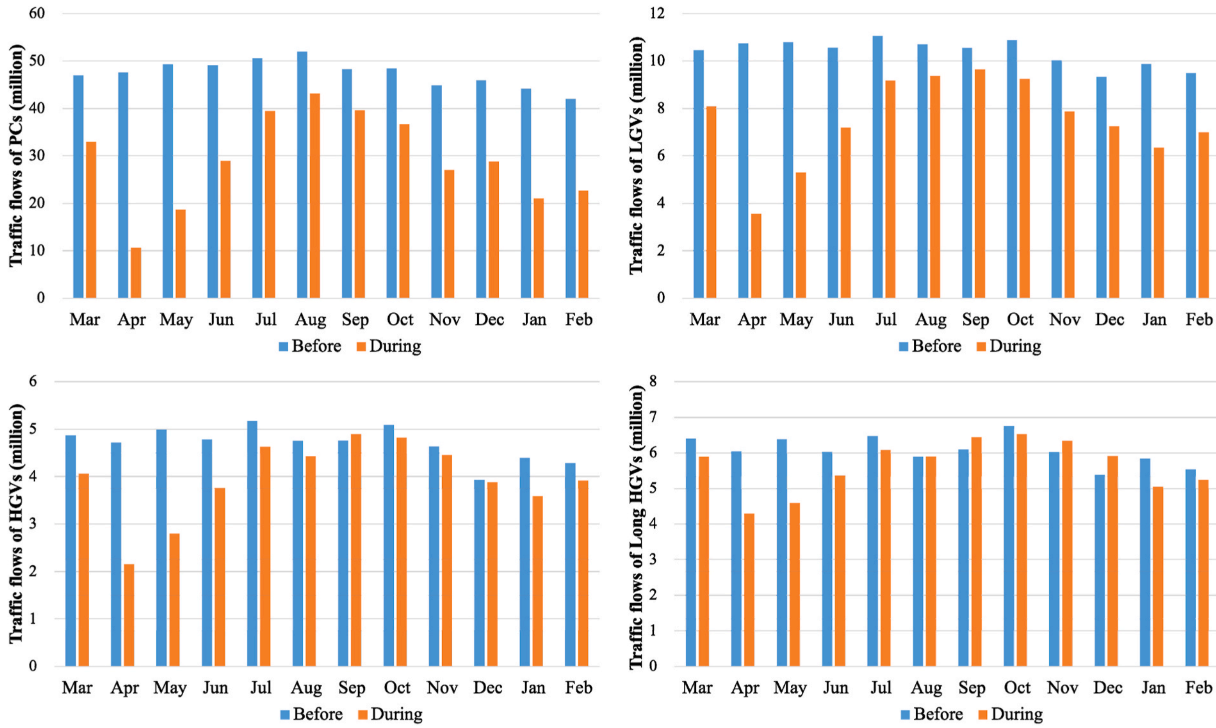

Fig. 3. Traffic flows of each vehicle category by month before and during the outbreak.

traffic flows of all vehicle categories increased again in February.

The percentage difference in traffic flows of four vehicle categories in the same months before and during the outbreak is illustrated in Fig. 4. Compared with April 2019, the traffic flow of four vehicle categories declined by 77.7%, 66.8%, 54.4%, and 29% in April of the first lockdown, respectively; the drop was the most significant throughout the outbreak. It can be seen that the reduction percentage in traffic flows for PCs in various months before and during the outbreak is the largest, followed by LGVs, HGVs and long HGVs. Compared to the year before the outbreak, it is interesting to note from Fig. 4 that the traffic flow for HGVs and long HGVs remains almost constant or even increases from September to December during the outbreak. This unexpected change in traffic flow is probably due to the fact that the food industry speeds up the development of online channels during COVID-19, thus leading to an increase in freight transport turnover (Ho et al., 2021; Liu et al., 2020; Mishra and Rampal, 2020).

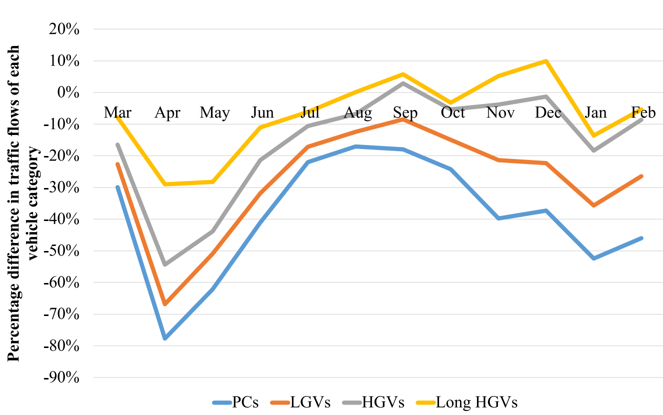

Fig. 4. Percentage difference in traffic flows of each vehicle category by month compared to pre-outbreak.

3.2. Non-exhaust emissions

The total mass of non-exhaust emissions per year was evaluated since the change in the trend of non-exhaust emissions from month to month was roughly consistent with traffic flow changes. Fig. 5 shows the total mass and proportion of $PM_{2.5}$ and PM_{10} in four types of non-exhaust emissions in the year before and during the outbreak. Compared to the year before the outbreak, although total $PM_{2.5}$ and PM_{10} mass decreased by 22.9% and 21.4% during the outbreak, respectively, the proportion of $PM_{2.5}$ and PM_{10} for each emission type remained nearly unchanged, with the difference being within 2%. Among the non-exhaust particles, the resuspension of road dust made the largest contribution to $PM_{2.5}$ and PM_{10} on the M25 motorway, followed by road wear, tyre wear, and brake wear emissions. In addition, the calculated results showed that the PM_{10} mass of non-exhaust emissions was 1.6 times larger than the $PM_{2.5}$ mass both before and during the outbreak.

3.2.1. Resuspension of road dust

Fig. 6 illustrates the total $PM_{2.5}$ and PM_{10} mass of road dust resuspension for each vehicle category before and during the outbreak. It can be seen that compared to the year before the outbreak, road dust resuspension of $PM_{2.5}$ and PM_{10} for the PC category reduced significantly during the year of the outbreak; the corresponding reduction in emissions for the long HGVs was the smallest. It was found from close observation that although the number of long HGVs was relatively small, their total road dust resuspension $PM_{2.5}$ and PM_{10} per year before and during the outbreak were the largest among the four categories of vehicles. Other researchers have reported similar findings. For example, Gillies et al. (2005) calculated PM_{10} emission factors of road dust resuspension for a range of vehicle types. The results showed that PM_{10} emission factors increased from 0.8 g km^{-1} h^{-1} for a light passenger car with a mass of 1200 kg to 48 g km^{-1} h^{-1} for a large military vehicle with a mass of 18000 kg. Schaap et al. (2008) modelled road dust emissions on paved roads in Europe and found that emission factors for HGVs were roughly nine times higher than those for LGVs. Amato et al. (2012) estimated the emissions factors of road dust resuspension for different types of vehicles on the motorways; they found that emissions factors for

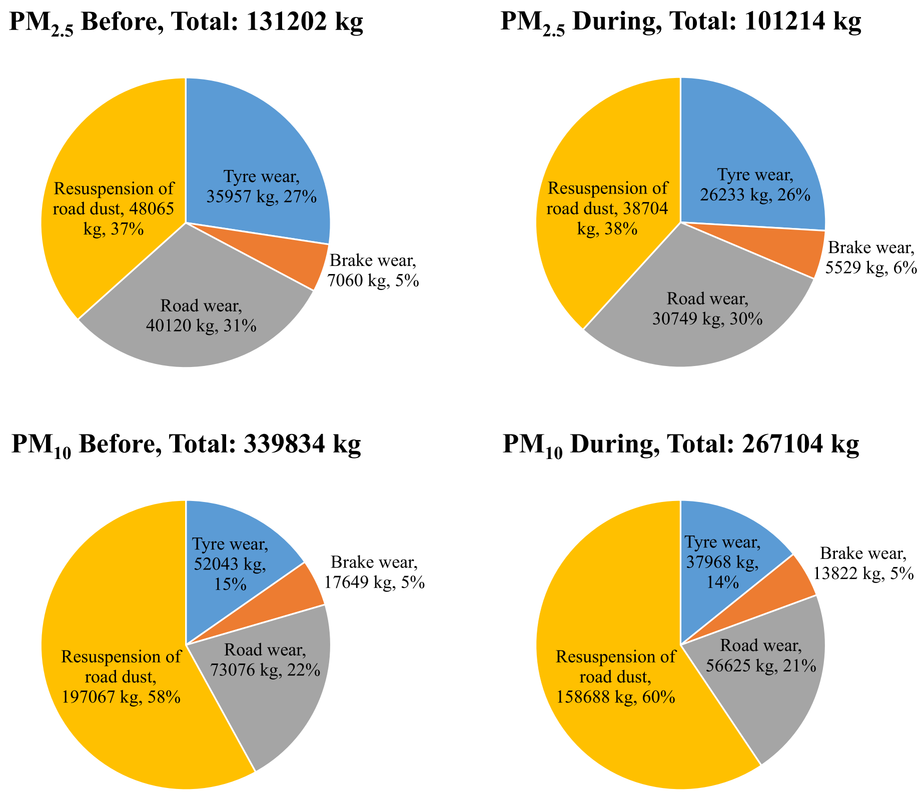

Fig. 5. The total mass and ratio of $PM_{2.5}$ and PM_{10} in different non-exhaust emissions before and during the outbreak.

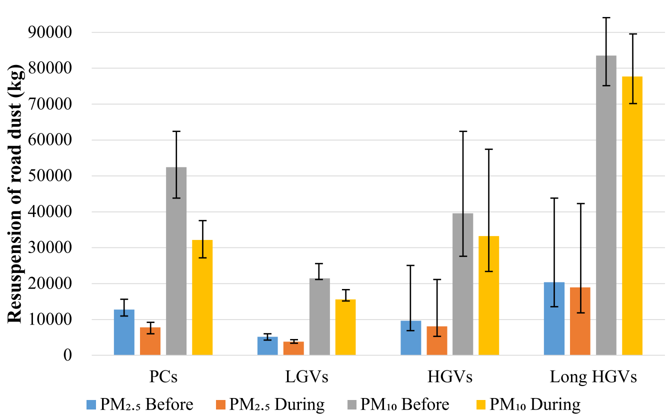

Fig. 6. Resuspension of road dust mass per year of each vehicle category before and during the outbreak.

heavy-duty vehicles were much larger than those for light-duty vehicles, PCs, and motorbikes. In addition, they found that the main components of the PM were mineral dust and organic and elemental carbon. However, OECD (2020) reported that it is controversial whether vehicle size or vehicle weight is responsible for this difference in road dust resuspension between small and large vehicles. They stated that aerodynamically, only vehicle size mattered, whilst heavier vehicle weight was likely to increase the tyre-induced lifting forces on deposited particles on roads, allowing more particles to be resuspended in the air. There is an alternative explanation for the large mass of road dust resuspension. According to the terminology and hierarchy put forward by Padoan and Amato (2018), this category of non-exhaust PM differs from direct wear emissions; it can contain brake, tyre, and road wear particles deposited on the road, as well as particles migrated to the road from other sources. Therefore, although the results of this study showed that road dust resuspension has the largest mass, the contribution of other types of non-exhaust PM emissions should not be neglected.

3.2.2. Road and tyre wear

Fig. 7 illustrates road wear $PM_{2.5}$ and PM_{10} masses per year before and during the outbreak for four main categories of vehicles on the M25 motorway. It can be observed that both $PM_{2.5}$ and PM_{10} masses from road wear of PCs were higher than that of other categories of vehicles in the year prior to the outbreak. This phenomenon is ascribed to the fact that the number of PCs is far higher than other vehicle categories, as demonstrated in Fig. 2. However, it is interesting that although the number of PCs was still higher than others, $PM_{2.5}$ and PM_{10} masses from road wear of long HGVs during the outbreak were the largest. The heavier weight of long HGVs may be primarily responsible for this. It is well known that road and tyre wear is caused by friction between the road surface and tyre tread, which is determined by both the normal force acting on the road as well as the friction coefficient between the tyres and the road (Timmers and Achten, 2016). This normal force is

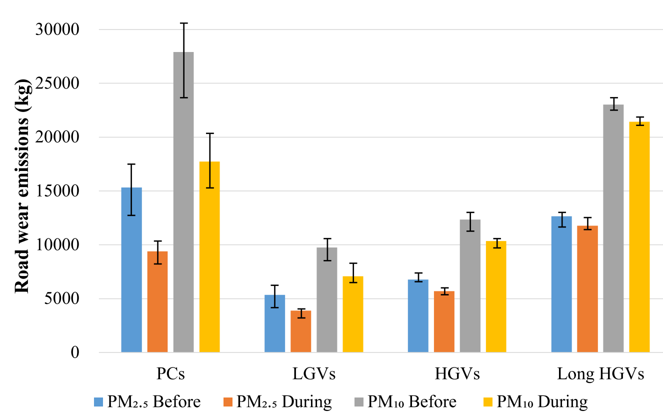

Fig. 7. Road wear emission mass per year of each vehicle category before and during the outbreak.

proportional to the vehicle weight, indicating that increased vehicle weight would enhance the friction force and thus increase road surface and tyre wear. A similar finding was reported by Žnidarič (2015), who pointed out that heavier axle loads for heavy-duty vehicles produced more road wear emissions.

Tyre wear $PM_{2.5}$ and PM_{10} masses per year before and during the outbreak for four main categories of vehicles on the M25 motorway is shown in Fig. 8. Unlike road wear, tyre wear $PM_{2.5}$ and PM_{10} masses per year before and during the outbreak were the largest for the PCs, nearly 1.3 times higher than those for the long HGVs. Compared to tyre wear $PM_{2.5}$ and PM_{10} masses per year from LGVs, long HGVs emitted more tyre wear emissions. From Fig. 2, the total number of LGVs per year before and during the outbreak was larger than that of long HGVs. However, the corresponding $PM_{2.5}$ and PM_{10} masses per year were less relative to those from long HGVs, as manifested in Fig. 8. It means that vehicle weight is a crucial factor affecting tyre wear emissions. Similarly, Andersson-Sköld et al. (2020), Li et al. (2012), and Wang et al. (2017) demonstrated that the loss mass of tyre wear showed an increasing trend with vehicle load. They ascribed this phenomenon to an increase in vertical contact pressure as vehicle load increased, resulting in increased slip forces and subsequent severe wear. An Emission Inventory of the Netherlands with emission factors for tyre wear by Ten Broeke et al. (2008) suggested that delivery vans emitted 40% more overall tyre wear $PM_{2.5}$ and PM_{10} emissions than the passenger car. In a report by Klein et al. (2014), it was found that tyre wear rates were 130% higher for light-duty vehicles than PCs and 20% higher for vans than for PCs.

3.2.3. Brake wear

Fig. 9 shows brake wear $PM_{2.5}$ and PM_{10} masses per year before and during the outbreak for four main categories of vehicles on the M25 motorway. Compared to the total mass of other non-exhaust particles per year, the total mass of brake wear emissions per year was the smallest. This is likely associated with the fact that speeds on motorways were generally constant, with less frequent acceleration and deceleration than urban roads (Yang et al., 2018). In contrast, Hicks et al. (2021) found that brake wear emissions were the largest contributor to non-exhaust $PM_{2.5}$ and PM_{10} emissions in the urban traffic of London. Such a difference in the contribution of brake wear to total non-exhaust emissions is likely associated with road conditions. Different road conditions significantly affect the brake frequency and thus give rise to variations in brake wear emissions. Beddows and Harrison (2021) and Liu et al. (2021) evaluated non-exhaust $PM_{2.5}$ and PM_{10} emissions from PCs on various roads; they found that brake wear emissions on urban roads were the largest, followed by rural roads and motorways. Previous studies have also demonstrated that high-frequency braking in urban roads and harsh braking would significantly increase brake wear emissions, especially for nanoparticles (Lee et al., 2013; Zum Hagen et al., 2019).

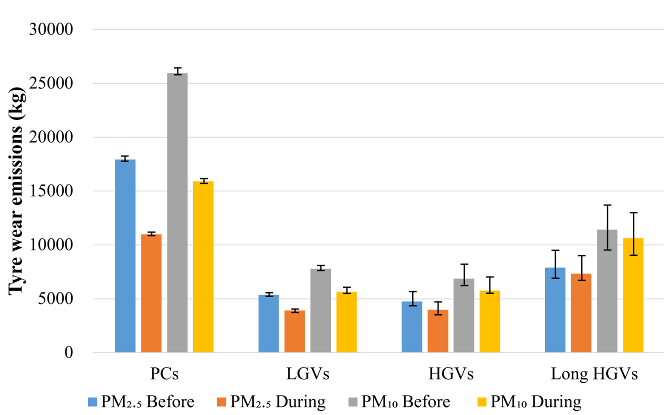

Fig. 8. Tyre wear emission mass per year of each vehicle category before and during the outbreak.

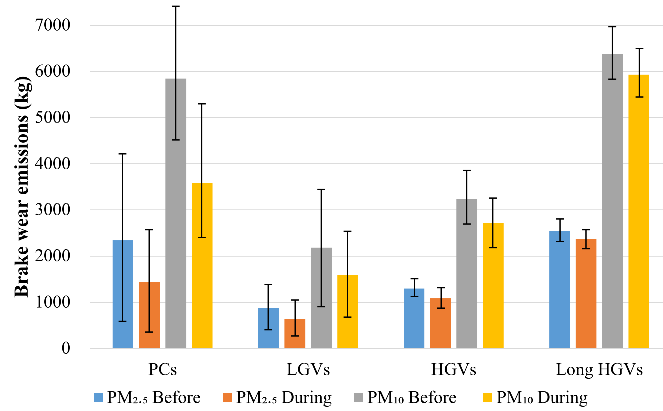

Fig. 9. Brake wear emission mass per year of each vehicle category before and during the outbreak.

From Figs. 2 and 9, although the total number of HGVs and long HGVs was relatively small on the M25 motorway, their brake wear $PM_{2.5}$ and PM_{10} masses per year before and during the outbreak were substantial, especially for long HGVs. It means that vehicle type significantly influences tyre wear $PM_{2.5}$ and PM_{10} emissions. Other researchers have reported similar views. For instance, Garg et al. (2000) reported that brake wear from large pickup trucks was more than twice that of small cars. Also, they found that larger PCs emitted 55% more total emissions, including $PM_{2.5}$ and PM_{10} related to small PCs. The U.S. Environmental Protection Agency (USEPA, 2014) distinguished brake wear $PM_{2.5}$ and PM_{10} emission factors from PCs and passenger trucks and demonstrated that the latter emitted 67% more PM_{10} and $PM_{2.5}$ brake wear emissions.

4. Conclusions

This work quantifies the impact of change in traffic flows of various categories of vehicles on the vehicle non-exhaust $PM_{2.5}$ and PM_{10} emissions on the M25 motorway before and during the outbreak. Compared to the year before the COVID-19 outbreak, the total traffic flows of PCs, LGVs, HGVs, and long HGVs reduced by 38.6%, 27.6%, 15.9% and 7.2%, respectively. Similarly, the corresponding total non-exhaust particles dropped by 38.7%, 27.3%, 16.2% and 7%, respectively. Among the non-exhaust particle emissions, resuspension of road dust on the M25 motorway was the largest contributor to air pollution, followed by road wear, tyre wear, and brake wear emissions. Moreover, it was found that the traffic flows of long HGVs were 87.2% less than those of PCs in the year prior to the COVID-19 outbreak, whilst the long HGVs emitted only 10.2% less $PM_{2.5}$ emissions but 10.9% more PM_{10} emissions than the latter. In parallel, the traffic flows of long HGVs were 80.7% less than those of PCs in the year of the COVID-19 outbreak, whilst the long HGVs emitted 36.3% more $PM_{2.5}$ emissions as well as 66.7% more PM_{10} emissions. These results showed that long HGVs contribute much more to non-exhaust particles than PCs. Regarding policy implications of the current empirical findings, a significant reduction in vehicle non-exhaust PM emissions would benefit from a traffic flow control on motorways. Automobile lightweight for all HGVs on motorways may be necessary for the UK in the coming future. There were a few limitations to this study. Only the representative M25 motorway was considered, which may limit the overall reflection of changes in traffic flows and corresponding non-exhaust particulate emissions on UK motorways. We also did not investigate vehicle exhaust emissions. Further quantification in these aspects to support more reliable predictions of the impact of traffic flow change on vehicle emissions is necessary and suggested for future research.

Author statement

Siyi Lin: Investigation, Data visualisation, Writing – original draft. **Ye Liu**: Investigation, Methodology, Writing – original draft, Writing – review & editing. **Haibo Chen**: Investigation, Funding acquisition, Project management. **Sijin Wu**: Investigation, Methodology, Software, Formal analysis. **Vivi Michalaki**: Conceptualization, Writing – review & editing. Phillip Proctor: Resources, Writing – review & editing. **Gavin Rowley**: Resources, Writing – review & editing.

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.

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Appendix A. Supplementary data

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