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Exhaust and non-exhaust airborne particles from diesel and electric buses in Xi'an: A comparative analysis

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Abstract: Switching diesel buses (DBs) to electric buses (EBs) has been a global trend to reduce the use of fossil fuels and improve air quality. However, buses electrification may lead to additional vehicle weight, which may emit more non-exhaust particulate matter (PM) emissions. It remains debatable whether buses' electrification will successfully improve air quality as excepted. To assess the effect of the buses’ electrification on the levels of PM emissions, PM emission factors (EFs) were evaluated from EBs and equivalent DBs. In addition, the total mass of PM emissions from EBs and equivalent DBs in 2021 was calculated in Xi’an using the real-world number and mileage of EBs. The non-exhaust PM EFs from EBs were larger than total exhaust and non-exhaust PM EFs from DBs, indicating that the electrification of buses would cause an increase in the level of PM emissions. The total annual mass of PM emissions from EBs was apparently higher than that from DBs. Moreover, a sensitivity analysis showed that tire wear, brake wear, and road wear PM emissions were more reliant on vehicle mileage, whereas resuspension of road dust was more dependent on vehicle weight. This finding can serve as a guideline for policymakers to design mitigation strategies for reducing extra PM emissions due to the electrification of buses by reasonably

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reducing vehicle weight and annual mileage.

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1. Introduction

Vehicle electrification, especially for buses, has been regarded as a solution to air pollution, with zero emissions and the promise of cleaner urban air (Tepanosyan et al., 2017; Bai et al., 2022). As a result, many policies continue to enhance the electrification of the bus fleet (Cooney et al., 2013; Zou et al., 2016; Al-Ogaili et al., 2021), which leads to a substantial increase in the number of electric vehicles. However, these advocates, perhaps, ignore non-exhaust particulate matter (PM) emissions from electric vehicles (i.e., tyre wear, brake wear, road wear, and dust resuspension) (Thunis et al., 2021; Liu et al., 2022a). Compared to conventional vehicles, the equivalent electric vehicles would produce more non-exhaust PM emissions due to the heavier battery pack compared to an equivalent conventional engine (Hooftman et al., 2016; Hong et al., 2020). In addition, previous studies have proven that non-exhaust emissions are a major source of PM emissions as a better understanding of the generation mechanism and physicochemical properties of PM improves the development of combustion control and after-treatment technologies, and thus vehicle exhaust PM is reduced significantly (Mor et al., 2021; Liu et al., 2016, 2021b; Zhang et al., 2021). For example, Rexeis & Hausberger (2009) revealed that non-exhaust emissions account for 90% of total PM emissions from traffic. Non-exhaust emissions consist of $PM_{2.5}$ and PM_{10}, which are small particles (the sizes below 2.5 micrometres diameter, the sizes from 2.5

to 10 micrometres diameter) generated during bus operation, composed of four vehicle-related sources, including tyre wear, brake wear, road wear, and resuspension. These smaller particles are liable to become airborne, which is one of the most dangerous pollutants to human health. Specifically, non-exhaust $PM_{2.5}$ emissions contributed more than 80% of the total traffic $PM_{2.5}$. In terms of PM_{10}, about 50–85% of the total traffic PM_{10} emissions originated from non-exhaust sources (Ketzel et al., 2007; Amato et al., 2014). This means that the switch from conventional vehicles to electric vehicles may not decrease total PM emissions or improve air quality (Soret et al., 2014). Timmers & Achten (2016) pointed out that the non-exhaust emissions from electric vehicles may exceed all PM emissions generated from equivalent internal combustion engine vehicles, including exhaust PM emissions. Beddows et al. (2021) and Liu et al. (2021a) revealed that the non-exhaust emissions generated from electric vehicles without regenerative braking were more than those from equivalent internal combustion engine vehicles. PM is one of the most dangerous pollutants to human health, with short-term and long-term health effects (Yang et al., 2019). Specifically, PM can cause serious harm to the respiratory system (Riedl and Diaz-Sanchez, 2005; Liu et al., 2022b).

So far, in response to national energy conservation and emission reduction policies, the electrification of the bus fleet in most large cities has been implemented (Dhar et al., 2017). To the best of our knowledge, however, there is no information in the existing literature on how the electrification of buses affects the overall level of PM emissions. In this context, the purpose of this study is to evaluate the effect of electrification on the overall level of PM emissions from EBs. The non-exhaust $PM_{2.5}$ and PM_{10}

emissions of EBs and equivalent DBs were calculated according to the method proposed by (Beddows et al., 2021). In addition, a comparative analysis of total PM mass from EBs and equivalent DBs in 2021 is performed according to the annual mileage data collected from the Xi'an traffic information center. Current results show that the electrification of buses would increase the mass of PM emissions. In addition, limiting the number of EBs batteries and optimizing bus mileage by replanning bus routes and rescheduling timetables would reduce non-exhaust emissions considerably.

2. **Methods**

Xi'an, as a typical city in China, was chosen to conduct this research. EBs in Xi'an are classified into two types: medium (between 6 m and 9 m) and large buses (between 9 m and 12 m) according to the People's Republic of China Transportation Standards JT/T 888-2020. In 2021, 5,335 EBs (4,871 large EBs and 464 medium EBs) operated with 250 routes in Xi'an. The average mileage of these EBs was 719,807 km (664,541 for large EBs and 55,266 for medium EBs).

To compare the total PM emissions for the switch from DBs to EBs, it was required to calculate non-exhaust and exhaust EFs from medium and large EBs and their corresponding DBs, respectively. Every weight of EBs and corresponding DBs were obtained from an internet database (https:/ /www.chinabuses.com//).

There are three steps to calculate non-exhaust EFs: (1) $PM_{2.5}$ EFs and PM_{10} EFs from medium and large EBs and their corresponding DBs were counted; (2) the relationships between these $PM_{2.5}$ EFs and PM_{10} EFs and the mass of various vehicle types were built; (3) the mathematical expressions was proposed between $PM_{2.5}$ EFs and PM_{10} EFs and bus weight, as shown in Eq. (1).

$$EF_i = b_i(w_i/1000)^{\frac{1}{c_i}} \tag{1}$$

where EF_i is the non-exhaust EFs of bus i (mg km^{-1} veh^{-1}), i is the index of buses, W_i is the weight of bus i, b_i (mg km^{-1} veh^{-1}) and c_i are factors of bus i, which are listed in Table 1.

Table 1. The value of factors used to fit the non-exhaust EFs vs. bus weight on urban road.

	$PM_{2.5}$		PM_{10}	
	b	c	b	c
Tyre wear	5.8	2.3	8.2	2.3
Brake wear	4.2	1.9	11.0	1.9
Road wear	2.8	1.5	5.1	1.5
Resuspension	2.0	1.1	8.2	1.1

Exhaust PM EFs from DBs were obtained by the approach (Brown et al., 2018). Based on the obtained exhaust and non-exhaust PM EFs and vehicle mileage per year, the total PM emissions per year from EBs and equivalent DBs in Xi'an were calculated using Eq. (2).

$$M = \sum_i (EF_i + E_i) q_i / 10\hat{}6 \tag{2}$$

Where M (kg) is the total mass of PM emissions per year, EF_i (mg km^{-1} veh^{-1}) is non-exhaust EFs, E_i (mg km^{-1} veh^{-1}) is exhaust EFs (the exhaust EFs from EBs is zero), q_i (km) is the annual mileage.

This study starts by inputting the weight of EBs and equivalent DBs in Xi'an. Then, PM EFs were calculated from EBs and equivalent DBs using the method proposed by (Brown et al., 2018). Then, we match the real-world mileage data with buses and calculate the mass of PM emissions. Finally, we compare the mass of $PM_{2.5}$/ PM_{10} with various bus weights and mileage to explore the relationship between bus weight/mileage and PM emissions. The flow chart of this study is shown in Figure 1.

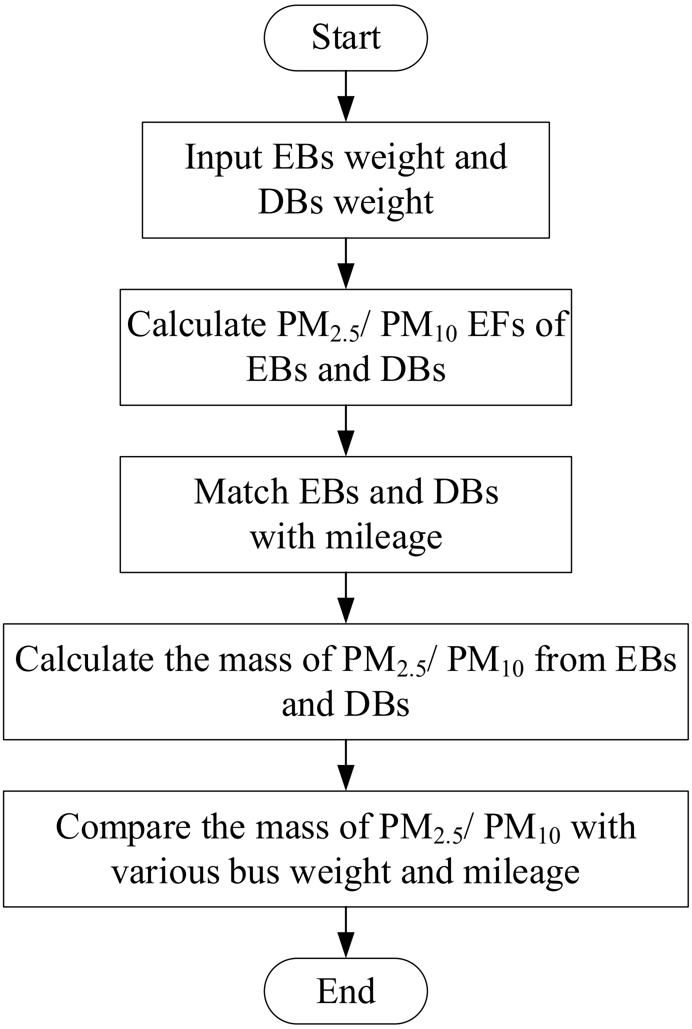

Figure 1. Flow chart of the proposed method.

3. Results and discussion

3.1. Weight assessment of buses

Each source of non-exhaust PM (i.e. tyre wear, brake wear, road surface wear, as well as resuspension) has a relationship with vehicle weight (Amato et al., 2011). Tyre and road wear are caused by friction which is strongly associated with the friction coefficient between them and the normal force against the road. Increasing vehicle mass leads to an increase in the normal force, thus emitting more non-exhaust emissions (Muller et al., 2003; Rajamani et al., 2010). Friction between the brake pad and disc produces brake wear particle emissions. Heavier vehicle weight requires more energy to reduce the vehicle kinetic energy, which increases brake wear particle emissions (Archard, 1953). Additionally, heavier vehicle weight is likely to increase the tire-induced lifting forces on deposited particles on roads, allowing more particles to be resuspended in the air (Alshetty and SM, 2022).

Previous studies have proved that heavier vehicles generate more non-exhaust PM emissions. For instance, Simons (2016) computed the PM_{10} EFs from cars and found that compared with a small car with 1200 kg weight, the PM_{10} EFs from a large car and medium car with 2000 kg and 1600 kg weight increased around 100% and 50%, respectively. Wang et al. (2017) investigated the relationship between vehicle weight and tyre wear and found that vehicle weight and tyre wear have an approximately linear relationship. Garg et al. (2000) reported that big vehicles and pickup trucks generated more than 55% and 100% of brake wear PM emissions than small cars.

The electrification of buses increases vehicle weight, and Table 2 shows a weight comparison between EBs in Xi'an and their equivalent DBs. As shown in Table 2, the average weight of EBs were 7,358 kg and 12,522 kg, as well as 5,660 kg and 10,879 kg from their equivalent DBs, which were 1,698 kg (30%), and 1,643 kg (15%) heavier than their equivalent DBs, respectively. Faria et al. (2012) suggested that the average mass of electric cars was 15% heavier than their equivalent internal combustion engine cars. Electric cars were reported to be 24% heavier on average than their equivalent internal combustion engine cars (Timmers & Achten, 2016). Beddows et al. (2021) revealed that the electrification of internal combustion engine cars resulted in a 21% increase in average vehicle mass. In this study, the average mass of EBs in Xi'an was 1,671 kg (20%) heavier than their equivalent DBs. This means that the increased percentage for the electrification of conventional passenger cars and buses was similar.

Table 2. Weight comparison between EBs and corresponding DBs (https://www.chinabuses.com//).

Bus type	Type model of EB	Type model of DB	Bus weight, EB (kg)	Bus weight, DB (kg)	Difference (kg)
Medium size	NJL6809EV2	NJL6769G5	7350	6500	850
	EQ6800CACBEV12	EQ6710CTV	6340	4250	2090
	SLK6819UBEVN1	SLK6759US55	7800	7000	800
	KLQ6800GEVN7	KLQ6729GE5	7500	5100	2400
	ZK6815BEVG18	ZK6729DG5	7800	5450	2350
Large size	CK6100LGEV2	LCK6105HGA	12200	10150	2050
	BYD6100LGEV9	LCK6105HGA	12200	10150	2050
	BYD6122LGEV1	HFF6120G04DE5	12500	10780	1720
	NJL6129BEV55	NJL6129G5	12800	11800	1000
	BYD6100LGEV3	LCK6105HGA	12500	10150	2350
	BYD6122B2EV2	HFF6120G04DE5	12650	10780	1870
	ZK6125BEVG55	ZK6120HG2	11400	10500	900
	BYD6100LSEV4	JS6111SHP	12600	12000	600
	JHC6120BEVG4	HFF6120G04DE5	12300	10780	1520
	CK6120LGEV	HFF6120G04DE5	13800	10780	3020
	NJL6129BEV55	NJL6129G5	12800	11800	1000

3.2. EBs and DBs PM emissions

3.2.1. Non-exhaust PM emissions

Figure 2 illustrates the total non-exhaust $PM_{2.5}$ and PM_{10} EFs for the EBs and equivalent DBs. It can be seen that both total non-exhaust PM EFs (including $PM_{2.5}$ and PM_{10}) for EBs were higher than those for the equivalent DBs. In terms of the switch from medium DBs to corresponding EBs, the total non-exhaust $PM_{2.5}$ and PM_{10} EFs increased by 1.18 and 1.20 times, respectively. Similarly, there is a 1.09 and 1.11 times increase in $PM_{2.5}$ and PM_{10} EFs for the large buses electrification. To further determine the impact of bus electrification on different types of non-exhaust emissions, the PM EFs from four types of non-exhaust emissions were calculated, and the results are shown in Figure 3.

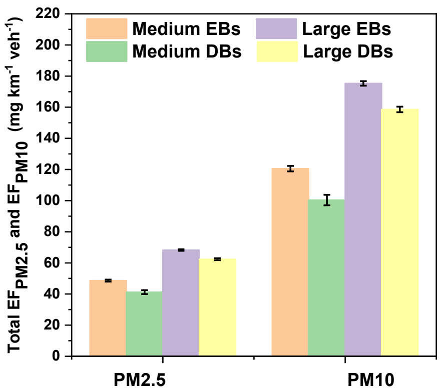

Figure 2. Total non-exhaust EFs for EBs-DBs including standard error.

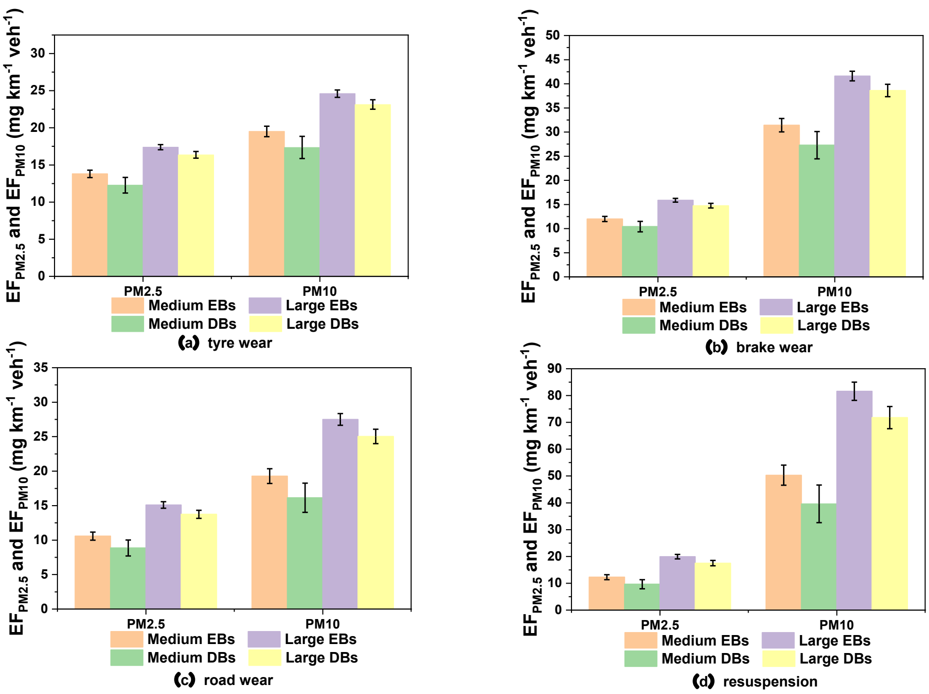

Figure 3. Non-exhaust EFs of EBs-DBs including standard error.

As seen in Figure 3, every type of $PM_{2.5}$ and PM_{10} EFs were all increased with the switch from EBs to equivalent DBs. Specifically, $PM_{2.5}$ EFs of tyre wear increased by 1.53 mg km^{-1} veh^{-1}, brake wear increased by 1.58 mg km^{-1} veh^{-1}, road wear increased by 2.72 mg km^{-1} veh^{-1}, and resuspension of road dust increased by 2.61 mg km^{-1} veh^{-1}, respectively, for the switch from large DBs to equivalent EBs. Correspondingly, PM_{10} EFs increased by 2.16 mg km^{-1} veh^{-1}, 4.14 mg km^{-1} veh^{-1}, 3.14 mg km^{-1} veh^{-1}, and 10.72 mg km^{-1} veh^{-1}, respectively, when switch from large DBs to equivalent EBs. Similarly, the electrification of medium buses also caused an increase in four types of non-exhaust emissions. A careful comparison revealed that the increased degree in each type of non-exhaust EFs for converting from medium DBs to equivalent EBs was larger than those for converting from large DBs to equivalent EBs. In a similar study, AQEG (2019) evaluated the non-exhaust PM emissions of EBs and DBs in the UK. However, it was found that four types of non-exhaust PM emissions from EBs were significantly higher than those in the present work. A possible explanation for this result may be that most buses in the UK are double-decker buses, which are heavier than single-decker buses in China, and heavier buses generate more non-exhaust PM emissions (Singh et al., 2020; Zhou et al., 2020). From Figure 3, the difference in PM EFs between medium EBs and large EBs was the largest for the resuspension of road dust, followed by brake wear, road wear, and tyre wear. It means that the PM EFs of road dust resuspension are sensitive to bus types. DBs show the same trend, and the reason is that the non-exhaust EFs for buses are dependent on bus weight, and the gap weight between large EBs and medium EBs is similar to this between large DBs and medium DBs.

In addition, the total non-exhaust PM mass per year was calculated from EBs in Xi'an and their equivalent DBs to explore the effect of buses' electrification on the overall emissions of non-exhaust PM emissions in a year. The calculated results for EBs and the equivalent DBs are listed in Table 3. From Table 3, the contribution of large-sized EBs to total non-exhaust PM in a year was much larger than that of medium-sized EBs, because most of the EBs in Xi'an are large EBs (i.e. there are 4,871 large EBs and 464 medium EBs, respectively). As a result, an effective solution to mitigate non-exhaust PM from large-sized EBs has the potential to significantly reduce non-exhaust PM from public transport in Xi'an.

Table 3. Comparative analysis of average non-exhaust PM mass from EBs to their equivalent DBs in Xi'an.

Non-exhaust PM	Types of buses	$PM_{2.5}$, EBs (kg)	$PM_{2.5}$, DBs (kg)	Difference (kg)	PM_{10}, EBs (kg)	PM_{10}, DBs (kg)	Difference (kg)
Tyre wear	Medium	288	258	30	406	364	42
	Large	4189	3900	289	5922	5513	409
Brake wear	Medium	249	219	30	653	573	80
	Large	3819	3502	317	10002	9173	829
Road wear	Medium	217	186	31	400	340	60
	Large	3625	3248	377	6602	5917	685
Resuspension	Medium	254	204	50	1040	835	205
	Large	4765	4105	660	19537	16830	2707
Total	Medium	1008	867	141	2499	2112	387
	Large	16398	14755	1643	42063	37433	4630

In terms of four types of non-exhaust emissions, it was found that the annual mass of resuspension PM_{10} emissions from large EBs was the largest, accounting for 58% of total non-exhaust PM_{10} mass in a year, followed by brake wear, road wear, and tyre wear. However, the order of the total annual mass of $PM_{2.5}$ emissions from large EBs was road dust resuspension, tyre wear, road wear, and brake wear. Compared to large DBs, the total mass of non-exhaust $PM_{2.5}$ and PM_{10} from equivalent large EBs in a year increased by 15% and 11%, respectively. Correspondingly, the annual $PM_{2.5}$ and PM_{10} mass from medium EBs rose by 14% and 10% more than the equivalent DBs, respectively. These results showed that the conversion from conventional DBs to equivalent EBs substantially increased the total annual mass of non-exhaust PM emissions. The result computed in this study is consistent with the result published in some studies (Zhang et al., 2018; Pan et al., 2019; Hicks et al., 2021).

3.2.2. Exhaust $PM_{2.5}$ and PM_{10}

Compared with EBs, DBs will generate exhaust PM emissions during operation due to diesel engine fuel (Maricq, 2007). In order to evaluate the impact of bus electrification on PM emissions, exhaust $PM_{2.5}$ and PM_{10} emissions from the DBs are also needed. The exhaust $PM_{2.5}$ EFs is 3.76 mg veh^{-1} km^{-1}, which is as same as PM_{10} EFs (Brown et al., 2018). Combining the fleet sizes and operating conditions in Xi'an, the annual exhaust $PM_{2.5}$ is equal to PM_{10} emissions (i.e. 69.95 kg). In the subsequent analysis, the authors compare PM emissions generated from DBs via exhaust and non-exhaust and PM emissions generated from EBs via non-exhaust to analyze extra PM emissions due to the electrification of buses.

3.3. Sensitivity analysis of total PM mass

A sensitivity analysis based on real-world data from Xi'an in 2021 was performed to explore the relationship between EBs weight and PM emissions. As shown in Figure 4, as the bus weight gradually decreased, the annual $PM_{2.5}$ and PM_{10} emissions reduced continuously. Specifically, the $PM_{2.5}$ and PM_{10} from EBs per year in Xi'an decreased by 551 kg and 1599 kg for every 5% reduction in EBs weight. This means that the light-weighting of EBs is probably an effective way to reduce non-exhaust PM emissions from public transport. The percentage variation in the total mass of annual PM emissions from conventional DBs to EBs with various vehicle weights was calculated, and the calculated results are shown in Figure 5. It was found in Figure 5 that the total mass of $PM_{2.5}$ from EBs without weight reduction or with 5% weight reduction was higher than those from corresponding DBs. For PM_{10}, the annual PM mass from EBs was more than those from corresponding DBs until reducing over 20% total weight of EBs. The results above indicate that the electrification of buses contributes more airborne particle emissions. As the bus weight decreases, the total amount of $PM_{2.5}$ emissions drops more quickly than PM_{10}, which may be attributable to the fact that compared to coarse PM_{10}, many factors, such as the vehicle weight and mileage had a more significant influence on fine $PM_{2.5}$.

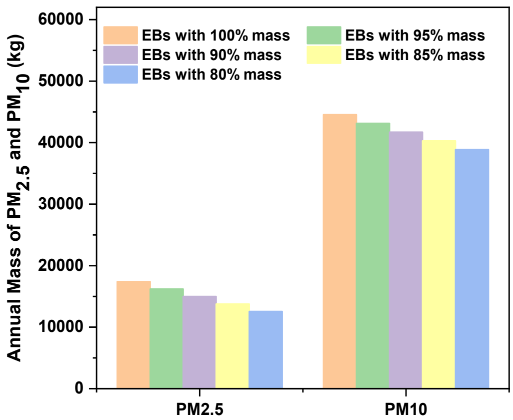

Figure 4. PM emissions for DBs and EBs with various weight.

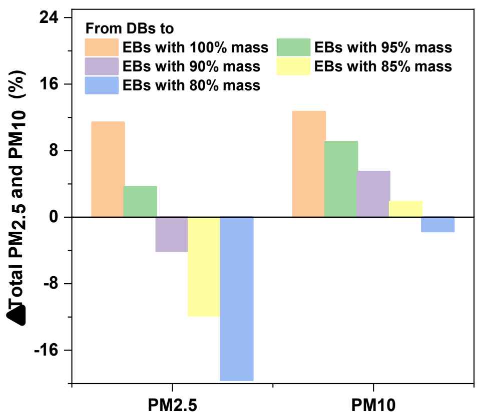

Figure 5. Percentage variation in PM emissions from DBs to EBs with various weight reductions.

In addition, a sensitivity analysis regarding the mileage and total PM mass was evaluated, and obtained results are shown in Figure 6. It was found that the annual mass of $PM_{2.5}$ and PM_{10} emissions from EBs in Xi'an was reduced by 892 kg and 2,230 kg when every 5% reduced the total mileage of EBs. Many previous studies have highlighted the need to optimize transit mileage from different perspectives. For instance, Wei et al. (2017) considered road conditions and the level of public traffic service to minimize the total mileage of buses. In addition, some studies optimize the total mileage of EBs to minimize total investment cost and CO_2 emissions (Ko et al., 2019). Compared to previous studies, this study provided a new application scenario that reduces the mass of PM emissions by optimizing the mileage of buses. Figure 6 illustrates the percentage variation of annual PM emissions from EBs with various mileage to conventional DBs. As shown in Figure 7, the mass of $PM_{2.5}$ and PM_{10} emissions from DBs will exceed those from EBs until reduced by 5% and 15% mileage, respectively. It also can find that compares to weight, adjusting the mileage can be more effective in reducing PM emissions in the public transport system of Xi'an. This finding is beneficial for policymakers in reducing the mass of $PM_{2.5}$ and PM_{10} emissions by replanning bus routes reasonably to minimize mileage with the premise of meeting traffic demand.

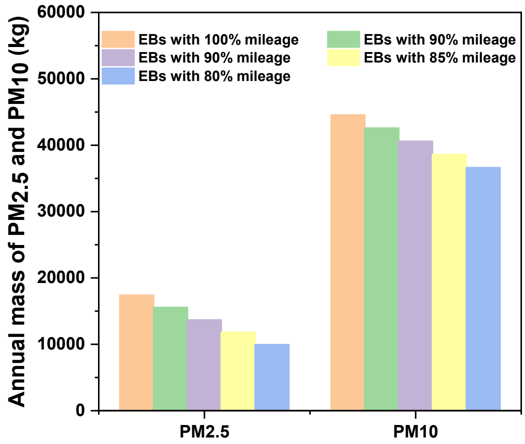

Figure 6. PM emissions for DBs and EBs with various mileage.

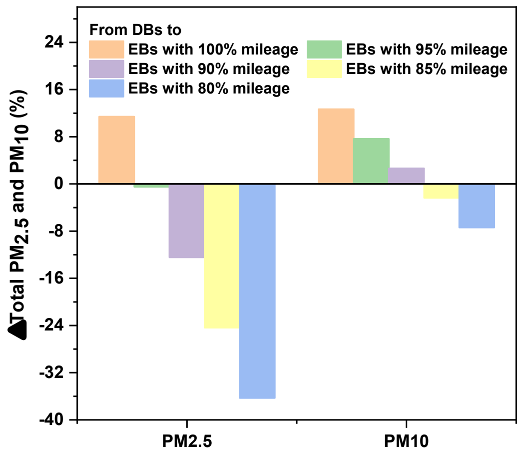

Figure 7. Percentage variation in PM emissions from EBs with various mileage and equivalent DBs.

In the present work, various non-exhaust emissions were evaluated with a 5% reduction in total vehicle weight and mileage using real-world data from the Xi'an bus operating system. The results are summarized in Table 4. Compared to a 5% reduction in total vehicle weight, the annual mass of the other non-exhaust PM emissions, except for the resuspension of road dust, decreased more when the total mileage was reduced by 5%. Specifically, compared to a 5% weight reduction, a 5% mileage reduction decreased 208 kg more $PM_{2.5}$ emissions from tyre wear, 9 kg more $PM_{2.5}$ emissions from brake wear, and 175 kg more $PM_{2.5}$ emissions from road wear, respectively.

Correspondingly, the annual mass of PM_{10} emissions was reduced by 296 kg, 179 kg, and 284 kg. By contrast, the total mass of dust resuspension emissions showed the opposite change. Compared to reducing 5% total mileage in Xi'an, the reduction of the mass of $PM_{2.5}$ and PM_{10} emissions was 1.21 and 1.12 times, respectively, when reducing 5% total bus weight. In other words, lowering weight is the more efficient strategy to minimize dust resuspension emissions. In the current work, only EBs in Xi'an were considered, so a more comprehensive analysis of other big cities is required to enrich the database. The data is beneficial for policymakers to make relevant strategies to mitigate non-exhaust PM emissions.

Table 4. The change of non-exhaust PM emissions from buses when reducing 5% total weight/ mileage.

	Reducing 5% total vehicle weight		Reducing 5% total vehicle mileage	
	$\triangle PM_{2.5}$ (kg)	$\triangle PM_{10}$ (kg)	$\triangle PM_{2.5}$ (kg)	$\triangle PM_{10}$ (kg)
Tyre wear	15	22	223	318
Brake wear	215	354	224	533
Road wear	18	66	193	350
Resuspension	303	1157	252	1029

4. Conclusions

In this study, the mass of PM emissions generated from different type of EBs and their equivalent DBs in Xi'an were evaluated using real-world bus fleets and mileage data in 2021. The current results indicate that the mass of $PM_{2.5}$ and PM_{10} emissions from EBs were more than those from equivalent DBs in Xi'an. The annual mass of PM emissions depends mainly on total bus weight and annual mileage. More specifically, every 5% reduction in bus weight and annual mileage will reduce 551 kg $PM_{2.5}$ emissions and 1,599 kg PM_{10} emissions as well as 759 kg $PM_{2.5}$ emissions and 1,894 kg PM_{10}, respectively. A sensitivity analysis revealed that resuspension of road dust was more dependent on vehicle weight. In contrast, tire wear, brake wear, and road wear PM emissions were more reliant on vehicle mileage. This finding is helpful for policymakers in cities to design the mitigation strategies to reduce extra PM emissions due to the electrification of buses by limiting the number of EB batteries when the EBs meet the mileage requirement and re-plaining the bus route reasonably to reduce mileage. As an extension of this study, the avenues of future research may include the following: (i) consider buses in other cities to enrich the data set; (ii) consider vehicle cost and emission cost values to propose practical strategies for optimizing non-exhaust emissions from EBs; (iii) analyze the variation of non-exhausted emissions under different stages (e.g., acceleration stage, low-speed stage or high-speed stage) to compute the PM emissions accurately.

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