TRL Limited

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ROAD VEHICLE NON-EXHAUST PARTICULATE MATTER: FINAL REPORT ON EMISSION MODELLING

Version: Final

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	NON-EXHAUST PARTICULATE MATTER EMISSIONS FROM ROAD TRAFFIC
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Quality Reviewed	*I McCrae*

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Executive summary

TRL Limited, the Division of Environmental Health & Risk Management (DEHRM) at Birmingham University and Cambridge Environmental Research Consultants Limited (CERC) have been commissioned by DEFRA to investigate non-exhaust particulate matter (PM) emissions from road traffic. The main aim of the project is to develop improved prediction methods for emissions and air pollution, primarily for use in the UK National Atmospheric Emissions Inventory (NAEI), based on the existing literature and data. The project is divided into five main Tasks:

Task 1: A literature review.
Task 2: Emission model evaluation, development and application.
Task 3: Initial air quality model development and application.
Task 4: Further air quality model development.
Task 5: Discussion of abatement options.

The findings of the literature review were presented in a previous TRL report. This report presents the findings of Task 2, the aims of which were to evaluate existing models for non-exhaust PM, to develop improved modelling approaches for use in the NAEI, and to apply these models to the UK. This second task was further broken down as follows:

Task 2a Model evaluation and development
2a(i) Emissions due to abrasion sources
2a(ii) Emissions due to resuspension
Task 2b Model application

Task 2a(i) involved the evaluation of existing PM emission models for abrasion sources - tyre wear, brake wear and road surface wear. The evaluation focussed on the EMEP, RAINS, CEPMEIP and MOBILE6.2 methods. In the absence of independent test data it was not possible to assess the absolute accuracy of the different approaches. The evaluation was therefore based upon a between-model comparison of emission factors for different size fractions and different vehicle categories. The EMEP method, which is currently used in the NAEI, is the most detailed approach, incorporating corrections for both speed and, in the case of HDVs, load and number of axles. It was concluded that none of the other three models or databases would currently offer any advantage over the EMEP method for application in the UK.

Task 2a(i) also involved the further development of emission models for abrasion sources. The abrasion model development phase of the work proceeded along the following lines:

- Further development of the EMEP method.
- Adaptation of the tyre wear model in HDM-4.
- Use of brake wear data from the US relating to PM emissions per braking event.

However, the lack of a substantial amount of new source-specific emission factors in the literature meant that no developments of the EMEP method were actually possible. Similarly, at present, brake wear data only exist for a limited number of braking conditions, and therefore the development of a prediction method according to this approach has not yet been possible. A spreadsheet-based version of the tyre consumption method in HDM-4 was compiled during the project, and a simple module was added to enable the prediction of PM_{10} emissions.

Five separate models were carried forward into Task 2a(ii), the determination of emission factors for resuspension. These models were:

- The existing EMEP method.
- The RAINS database.
- The CEPMEIP database.
- The HDM-4 model for tyre wear (combined with a PM_{10} emission calculation routine).
- A German 'traffic situation model' (used to predict total non-exhaust PM_{10} emissions).

In Task 2a(ii), the EMEP, RAINS and CEPMEIP methods were used by DEHRM to estimate emissions due to resuspension at particular localities, and specifically for Marylebone Road in London. The work was based mainly upon the EMEP method, with the RAINS and CEPMEIP databases being used to provide an indication of the sensitivity of the abrasive source estimate to the model used. The HDM-4 tyre wear model was also used to assess the effects of applying a detailed model to calculate tyre wear emissions. Although there may be concerns about the representativeness of the German traffic situation model for the UK, it was also considered appropriate to include this model for further sensitivity testing.

For Marylebone Road, and based on the EMEP emission factors, comparisons between the estimated resuspended component with total non-exhaust emissions suggested that resuspension accounts for 43-49% of the non-exhaust emissions. Furthermore, resuspension emissions were found to be around 30% of the magnitude of exhaust emissions. HDVs were found to be almost entirely responsible for the resuspension of particles. Emission factors for resuspension due to HDVs ranged from 139 mg/vkm to 145 mg/vkm. Much lower emission factors were observed of LDVs.

There was found to be a decrease in emissions of resuspended particles in the $PM_{2.5-10}$ size range between 2000 and 2002, due in part to the use of a constant $PM_{2.5-10}$:PM_{10} emissions ratio of 0.4. However, the use of observed $PM_{2.5-10}$:PM_{10} ratios to calculate $PM_{2.5-10}$ emissions, and subsequently to estimate resuspension, offered little improvement over the use of the constant ratio. Furthermore, the anticipated increase in the ratio $PM_{2.5-10}$:PM_{10} was not conclusively supported by data from Marylebone Road or the roadside increment data. It is also apparent that the selection of appropriate background sites for calculation of roadside incremental concentrations is imperative.

The estimates of abrasion emissions appeared to be quite sensitive to the method adopted. Nonetheless, the calculated values of resuspension were of a reasonable magnitude, and their variation with wind speed was broadly consistent with that which would be anticipated.

There was evidence of an increase in resuspension with wind speed, with the rate of increase apparently slowing as wind speeds get larger. It was postulated that the key role of the vehicle is in resuspending the particulate matter from the road surface initially, be it by tyre shear or vehicle-generated turbulence. The role of wind speed is in generating the more extensive atmospheric turbulence responsible for keeping the larger particles resuspended such that they then have a significant influence upon airborne concentrations. Analysis of the Marylebone Road data revealed no clear relationship between resuspension and precipitation.

So far, this study has shown that there are few detailed methodologies for predicting emissions of particulate matter from non-exhaust sources. Furthermore, there has been insufficient information presented in the recent literature to enable the further development of existing models from a source perspective. There is clearly a need for more extensive empirical data, and a number of general recommendations for methods of obtaining such data are made. It also clear from the study that significant quantitative insights can be gained from analysis of measured data from a heavily instrumented monitoring site, such as the one at Marylebone Road. However, one weakness of the current study is that it is based purely on a single street canyon site, and it is important to test the extent to which the results can be generalised to other locations having different characteristics.

1 Introduction

1.1 Overview

TRL Limited, the Division of Environmental Health & Risk Management (DEHRM) at Birmingham University and Cambridge Environmental Research Consultants Limited (CERC) have been commissioned by DEFRA to investigate non-exhaust particulate matter (PM) emissions from road traffic. The main aim of the project is to develop improved prediction methods for emissions and air pollution, primarily for use in the UK National Atmospheric Emissions Inventory (NAEI), based on the existing literature and data. The project is divided into five main Tasks:

Task 1: A literature review.
Task 2: Emission model evaluation, development and application.
Task 3: Initial air quality model development and application.
Task 4: Further air quality model development.
Task 5: Discussion of abatement options.

The findings of the literature review were presented in a previous TRL report (Boulter, 2005a). The review summarised the available information on particle emissions from road vehicle non-exhaust sources, including the methodologies employed to measure and model emissions, and provided recommendations for model development during the remainder of the project. The conclusions drawn from this review, including some specific recommendations for model development, are summarised in Section 1.2.

The aim of Task 2 of the project is to evaluate the existing models for non-exhaust PM, to develop improved modelling approaches for use in the NAEI, and to apply these models to the UK. This second task can be further broken down as follows:

Task 2a Model evaluation and development
2a(i) Emissions due to abrasion sources
2a(ii) Emissions due to resuspension

Task 2b Model application

This Interim Report presents the findings of Task 2, the evaluation, further development, and application of emission models. In Task 2a(i) models for the abrasion sources - tyre wear, brake wear and road surface wear – were considered separately. Prediction methods for total emissions of non-exhaust particles were also considered. In Task 2a(ii), a number of different emission modelling methods were used to develop an improved method for estimating emissions due to resuspension at specific locations. Task 2b involved the calculation of non-exhaust PM emissions in the UK, based on a revised set of emission factors. A glossary, explaining the terminology used in this report, is provided in Appendix A.

1.2 Conclusions and recommendations of literature review

1.2.1 Conclusions

The following conclusions were drawn from the literature review in Task 1 (Boulter, 2005a):

1. The most important non-exhaust sources of airborne PM are likely to be tyre wear, brake wear, road surface wear and resuspension. For UK locations, the relative importance of the different sources is highly uncertain.

2. There is a lack of consistency in the terms, definitions and metrics used in the study and reporting of non-exhaust PM. For example, sources may be reported independently or in combination. This often hinders both the incorporation of data into models and the comparison of model predictions with earlier studies. The various sources need to be considered as independently as possible, and causal relationships need to be identified.

3. For the tyres of light-duty vehicles (LDVs) under 'normal' driving conditions, a typical wear factor[1] is 100 mg/vkm[2]. For heavy-duty vehicles (HDVs), the wear factor is likely to be an order of magnitude higher. In the case of LDVs, probably less than 10% of tyre wear material is released as PM_{10}; no corresponding estimates are available for HDVs.

4. The brake wear factors for LDVs and HDVs appear to be around 10-20 mg/vkm and 50-80 mg/vkm respectively. For LDVs, typically 50% of the brake wear PM escapes the vehicle and enters the atmosphere. More than 80% of the airborne particles can be classified as PM_{10}, with a substantial $PM_{2.5}$ fraction. However, there is considerable uncertainty regarding the amount of material which is lost from the brake linings, and the amount which is lost from the discs or drums.

5. A wide range of road surface wear factors have been reported, from less than 4 mg/vkm to more than 400 mg/vkm. However, in areas of northern Europe where there is extensive use of road sanding and studded tyres during the winter, the wear of the road surface is considerably higher - values of between 4 and 24 g/vkm have been reported. Very little information on the size distribution of road surface wear particles is available.

6. Resuspension is probably the single largest vehicle non-exhaust contributor to roadside PM_{10}, particularly where winter maintenance procedures are in place. Although some information on the effects of winter maintenance is available for Nordic countries, the situation in the UK is rather different. In the UK, road de-icing involves the application of rock salt, but little information of the effects of this particular approach on resuspension has been reported. This specific subject requires further investigation.

7. Models for non-exhaust PM emissions are generally rather crude, and more detailed methodologies are required. The use of average brake wear emission factors (in g/km) in models does not seem particularly logical, as the differences in the extent of braking for different traffic situations cannot be taken into account.

8. Emission factors for resuspension are variable, and most of the available information is derived from sites in the United States, where the conditions associated with measurement appear to have often been dry and dusty, or in Nordic countries where the use of studded tyres presents a significant problem. Total PM_{10} emission factors in the US generally appear to be substantially higher than those in Europe. US prediction models, such as the AP-42 model for paved roads are therefore unlikely to be appropriate to the UK.

1.2.2 Recommendations for emission model development

In the literature review, it was recommended that consideration should be given to the following aspects of emission modelling within Task 2 of the project:

1. The acquisition of further information on the non-exhaust PM emission models covered in the review, and any other relevant models.
2. The acquisition of any data relating to UK conditions which were required to run the models.

[1] In this context, the 'wear factor' is defined as the total amount of material lost to the environment (in g/km).
[2] vkm = vehicle-kilometre.

3. The evaluation of model performance for a range of typical UK conditions, using real-world data where possible.
4. The selection of several models for further refinement and application. It was expected that different modelling approaches would be required for direct emission sources and resuspension.
5. The possibility of combining brake wear emission factors stated in terms of mass per stop and driving pattern data for different UK traffic situations.

In order to improve the understanding of non-exhaust PM in the UK, a number of general recommendations were also provided (Boulter, 2005a).

1.3 Report structure

Chapter 2 of the Report contains a basic evaluation of existing PM emission models for tyre, brake and road surface wear. Chapters 3 describe the further development, where possible, of these models, and lists a number of alternative approaches. The results of the investigations into resuspension are presented in Chapter 4, and Chapter 5 describes the estimation of non-exhaust PM emissions in the UK. Chapter 6 provides the summary, conclusions and recommendations from the work conducted so far.

2 Evaluation of existing models

2.1 Model identification

Task 2a(i) concerned the evaluation and further development of models for predicting PM emissions from abrasion sources, and focused on models which treat non-exhaust particulate matter from a source perspective (*i.e.* they describe emissions), as opposed to models which describe it from a receptor perspective (*i.e.* they involve source apportionment of ambient PM).

The first stage of Task 2a(i) was the identification of existing models, for which the following sources of information were considered:

(i) The review by Boulter (2005a), which identified the following models for abrasion sources:

- EMEP/CORINAIR Emissions Inventory Guidebook
- RAINS
- CEPMEIP
- MOBILE 6.2

and the following models for predicting total non-exhaust PM emissions:

- USEPA[3] AP-42 model for paved road dust
- Modified AP-42 for use in Germany
- German traffic situation model
- SMHI model
- VLUFT model

(ii) Further searches of publication databases and the internet.

(iii) Direct approach to tyre manufacturers:

- Bridgestone
- Continental
- Dunlop
- Goodyear
- Michelin
- Pirelli
- Yokohama

(iv) Direct approach to brake manufacturers:

- Bosch
- Delphi
- Ferodo
- Mintex

(v) Direct approach to model developers and researchers:

- European Commission TROWS project (TNO, CETE, Politecnico di Milano, Infralab)
- HDM-4 (University of Birmingham, World Bank)
- Ford
- VTI

Other than the models described by Boulter (2005a), no other methodologies for modelling PM emissions from abrasion sources were obtained. The model evaluation for abrasion sources therefore focussed on the EMEP/ CORINAIR Emissions Inventory Guidebook, RAINS, CEPMEIP and MOBILE6.2. These models are described briefly in Section 2.2, and the evaluation is presented in Section 2.3.

[3] United States Environmental Protection Agency.

Approaches to tyre and brake manufacturers, to other research institutions, and to model developers yielded little information which could be used directly in the project. However, one other modelling approach was considered to be directly relevant to this project - the mechanistic model used for predicting tyre wear in HDM-4. The model of Patra *et al*. (2005) from the DAPPLE[4] project for predicting traffic-derived airborne PM was also given consideration, but it was felt that there would be insufficient time to obtain the input data to allow the model to be adequately tested and applied. These approaches are described in Chapter 3 of the Report.

2.2 Model descriptions

The available models or databases which include information on direct PM emissions due to tyre, brake and road surface wear are:

- The model in the EMEP/CORINAIR Emissions Inventory Guidebook (EEA, 2004) – hereafter referred to as the '**EMEP**' method. EMEP presents a methodology for determining primary non-exhaust PM emissions which was developed by TRL and LAT. It has also been used in the latest revision to the NAEI (NETCEN, 2005). The methodology covers tyre, brake and road surface wear; resuspension is not included. Two methods are described, a simple method for PM_{10}, and a detailed method for various size metrics which includes corrections for speed and HDVs size and load (Appendix B). The detailed method was used in this study.

- The database of the **RAINS** model. The European Commission has established the 'Clean Air For Europe' (CAFE) programme to conduct a systematic review of all EU legislation relating to air quality. The Commission has decided that the RAINS model (Alcamo *et al.* 1990), developed by IIASA, will serve as the central integrated assessment tool for CAFE.

- The **CEPMEIP**[5] database. CEPMEIP supports national experts in reporting particulate matter emission inventories to the UNECE Convention on Long Range Transport of Air Pollutants (CLRTAP). The emission factors used in CEPMEIP for tyre, brake and road surface wear are derived from the international literature (Berdowski *et al.,* 2001).

- The USEPA emission factors currently used in the **MOBILE 6.2** model.

These models and databases were described by Boulter (2005a), and they are evaluated in this Chapter of the Report. At this stage, it must be appreciated that only a relatively small number of PM emission measurements are available for abrasion sources, and these data are rather variable. Although most of the existing information has been included in the above methodologies to a greater or lesser extent, each model developer will have used the information in a different way and for different reasons. Not surprisingly, this has led to a general lack of agreement in the emission factors used in different models. The sources of data used in the four models, where available, are shown in Table 1, though not all the listed sources refer to primary data.

4 http://www.dapple.org.uk/

5 Co-ordinated European Programme on Particulate Matter Emission Inventories, Projections and Guidance.

Table 1: Sources of data used in EMEP, RAINS, CEPMEIP and MOBILE 6.2.

Model/ database	Tyre wear			Brake wear			Road surface wear
	MC	LDV	HDV	MC	LDV	HDV	
EMEP†	Gebbe *et al.* (1997) Garben *et al.* (1997) Env. Austral. (2000)	USEPA (1985b) Gebbe *et al.* (1997) Garben *et al.* (1997) Lee *et al.* (1997) Baumann *et al.* (1997) Rauterberg-Wulff (1999) Legret *et al.* (1999) Koliousis *et al.* (2000) EMPA (2000) Luhana *et al.* (2004)	Baumann *et al.* (1997) Gebbe *et al.* (1997) Rauterberg-Wulff (1999) Legret (1999)	BUWAL (2001)	Cha et al. (1983) Legret *et al.* (1999) Garg *et al.* (2000) Westerlund (2001) Luhana *et al.* (2004)	Rauterberg-Wulff (1998) Legret *et al.* (1999) Westerlund (2001)	Lükewille *et al.* (2001) (RAINS)
RAINS	Env. Austral. (2000) EMPA (2000)	USEPA (1985a) Rauterberg-Wulff (1998) EMPA (2000)	USEPA (1985a) Rauterberg-Wulff (1998) EMPA (2000)	BUWAL (2001)	BUWAL (2001) Rauterberg-Wulff (1998) Garg *et al.* (2000) Env. Austral. (2000) USEPA (1985a) Cha *et al.* (1983)	BUWAL (2001) Rauterberg-Wulff (1998) Garg *et al.* (2000)	Based on total non-exhaust PM minus PM due to tyre wear, brake wear and resuspension : Israel *et al.* (1994) Israel *et al.* (1996) Berdowski *et al.* (1997) CBS (1998) Rautenberg-Wulff (1998) EMPA (2000)
CEPMEIP	Dreiseidler *et al.* (1999) Klimont *et al.* (2002) (RAINS) Klein *et al.* (2003) USEPA (1985a) MEET/COST319 data (not specified)			Dreiseidler *et al.* (1999) Klimont *et al.* (2002) (RAINS) Klein *et al.* (2003) USEPA (1985a)			Dreiseidler *et al.* (1999) Klimont *et al.* (2002) (RAINS) Klein *et al.* (2003)
MOBILE 6.2	USEPA (1985b)			USEPA (1985b)			Not included

† A number of other references were considered for inclusion in the EMEP method. However, if reported values were particularly high or low they were not included.

2.3 Between-model comparison

In the absence of independent test data it was not possible to assess the absolute accuracy of the different approaches. The evaluation was therefore based upon a between-model comparison of emission factors for different size fractions and different vehicle categories. The size fractions considered in most detail were PM_{10} and $PM_{2.5}$, for which all models provide information. PM_1 and $PM_{0.1}$ were examined in less detail.

Because vehicle categories are defined in different ways in different models, a standardised nomenclature was used. Eight vehicle categories were defined for the purpose of this comparison:

- V1 = Two-wheel vehicles
- V2 = Cars
- V3 = Light goods vehicles (LGVs)
- V4 = HGVs, buses and coaches with 2 axles
- V5 = HGVs, buses and coaches with 3 axles
- V6 = HGVs with 4 axles
- V7 = HGVs with 5 axles
- V8 = HGVs with 6 axles

This categorisation was based primarily on that used in the most detailed model (EMEP). However, as not all models could distinguish between different vehicle categories at this level of detail, a number of assumptions were required:

- Both the EMEP method and MOBILE 6.2 allow emission factors to be specified for different types of HDV. However, in the case of tyre wear the EMEP method differentiates between HDVs according to number of axles, whereas MOBILE uses number of wheels. The assumed correspondence between axle number and wheel number for all types of vehicle is shown in Table 2. Estimated weight ranges and average weights are also given, as these are used later in the Report.

Table 2: Number of axles, number of wheels, estimated weight range and estimated average weight for the eight vehicle categories.

Vehicle category		Number of axles	Number of wheels†	Estimated weight range (t)	Estimated average weight (t)‡
V1	Two-wheel vehicles	2	2		0.2
V2	Cars	2	4	<= 2.5	1.2
V3	LGVs	2	6	<= 3.5	3
V4	HGVs, buses and coaches	2	6	3.5 to 18	8
V5	HGVs, buses and coaches	3	10	14 to 32	20
V6	HGVs	4	14	14 to 40	25
V7	HGVs	5	18	28 to 40	30
V8	HGVs	6	18	28 to 44	35

† For HDVs, assumed to be (4 x number of axles)-2
‡ Not based on published data

- In both RAINS and CEPMEIP, no distinction is made between different types of HDV. It is assumed that the emission factors are biased towards mid-range vehicles.

- Only the EMEP models refers specifically to buses, although it is assumed that the emission factors for 2-axle and 3-axle buses and coaches are the same as those for the equivalent HGVs.
- The EMEP method features a vehicle load term for HDV tyre and brake wear, having a value between 0 (empty) and 1 (fully loaded). For this comparison a value of 0.5 was used for both tyre and brake wear.
- No reference is made in any model to potential differences between petrol-engined and diesel-engined vehicles, and so it is assumed that the emission factors apply to vehicles using any fuel.
- RAINS does not distinguish between cars and LGVs, and so the same emission factors have been used for both.

2.3.1 Tyre wear

Figures 1 and 2 show the tyre wear emission factors used in the four models for PM_{10} and $PM_{2.5}$ respectively. The emission factors are plotted as a function of speed for the eight vehicle categories listed above. For HDVs an upper speed limit of 100 km/h was applied. The following observations have been made from these comparisons:

- EMEP incorporates a speed dependency, but only between 40 km/h and 90 km/h. Emission factors at lower and higher speeds are assumed to be constant, and it is therefore possible that the method does not account for some driving conditions which result in high emissions. However, the other models offer even less to the user in terms of defining vehicle operation, and simply use single emission factors for different vehicle categories and PM size fractions.
- For any given vehicle category, the different models can produce substantially different emission factors, reflecting the earlier comments concerning the constitution of the respective databases.
- For PM_{10}, the EMEP method tends to produce the highest emission factors for LDVs and large HDVs at low speeds (with a vehicle load factor of 0.5). For smaller HDVs, RAINS produces the highest emission factors.
- The $PM_{2.5}$ emission factors in the EMEP method are substantially higher at all speeds than those in the other three models. The CEPMEIP model assumes that no tyre wear particles are emitted in the $PM_{2.5}$ fraction.

However, it should be noted that recent unpublished research by TNO (the developers of CEPMEIP) indicates provisional tyre wear PM_{10} emission factors for LDVs and HDVs of 7 mg/vkm and 50 mg/vkm respectively, with a $PM_{2.5}$ fraction of 40% ± 20% (Visschedijk, 2005). For PM_{10} this leads to emission factors close to those in RAINS, and for $PM_{2.5}$ closer to the EMEP values.

The EMEP method also provides emission factors for PM_1 and $PM_{0.1}$. The MOBILE 6.2 method provides an emission factor for $PM_{0.1}$, but nothing for PM_1. The $PM_{0.1}$ values in MOBILE 6.2 are considerably lower than those in the EMEP method. Figure 3 shows and example of the EMEP emission functions for different size fractions, with the $PM_{0.1}$ value in MOBILE 6.2 included for comparison.

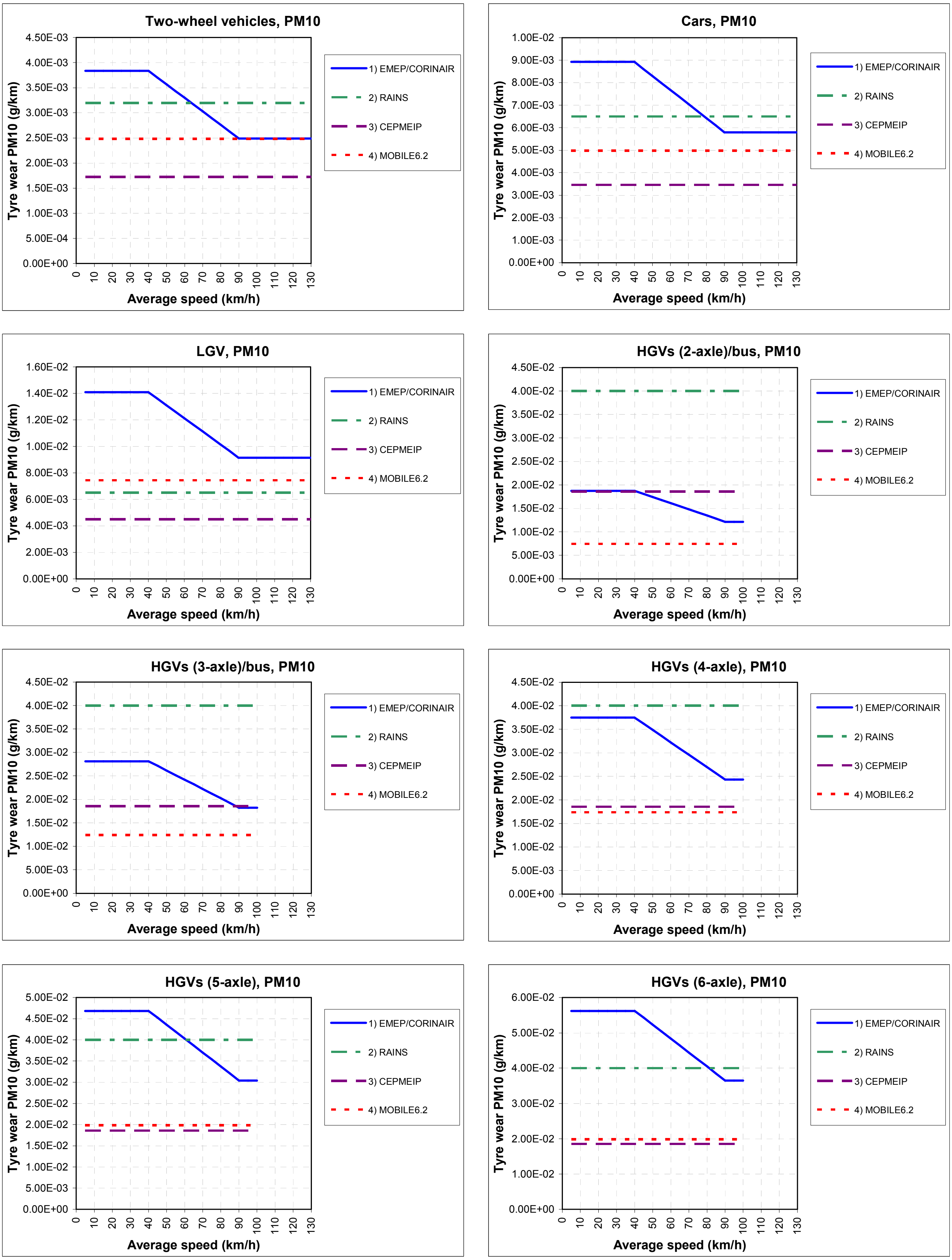

Figure 1: PM_{10} emission factors for tyre wear by vehicle type and emission model.

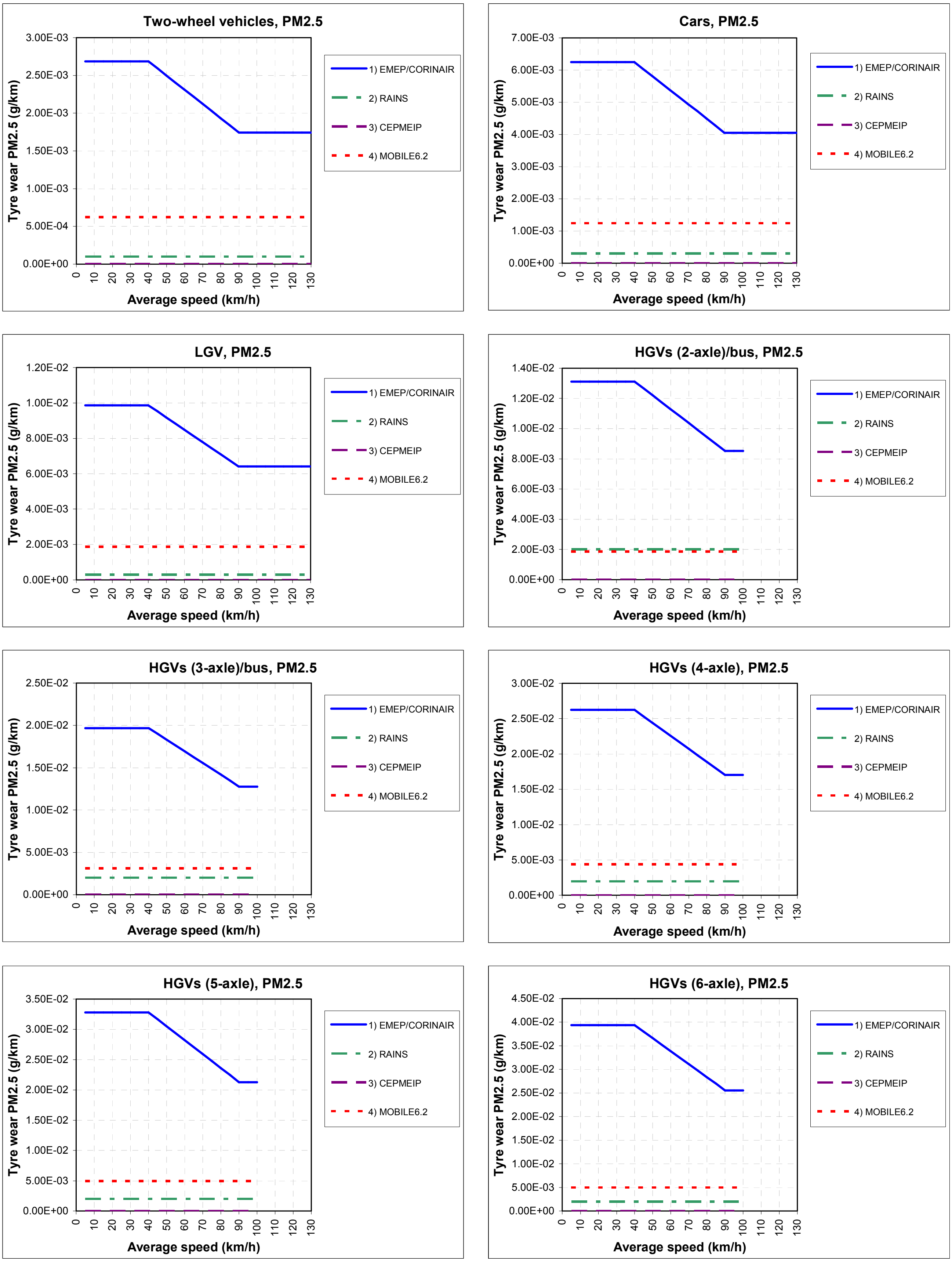

Figure 2: $PM_{2.5}$ emission factors for tyre wear by vehicle type and emission model.

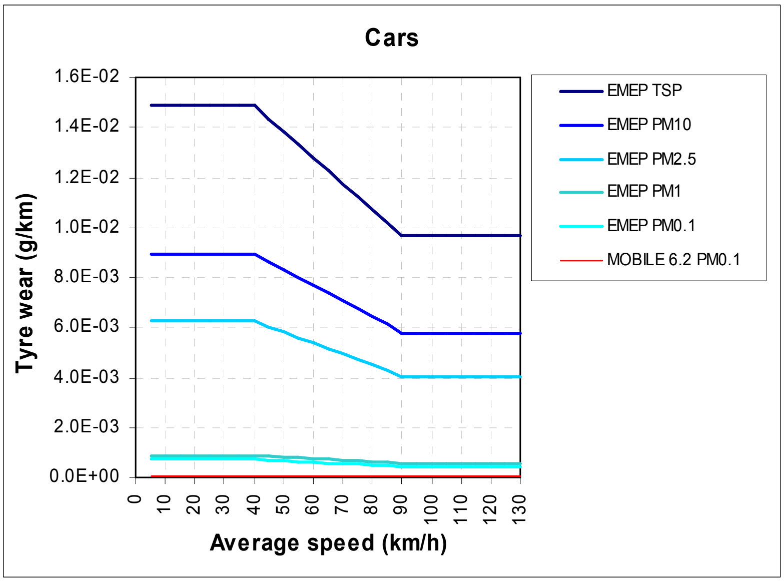

Figure 3: PM emission factors for car tyre wear by size fraction.

2.3.2 Brake wear

Figures 4 and 5 show the brake wear emission factors used in the four models for PM_{10} and $PM_{2.5}$ respectively. Again, the emission factors are plotted as a function of speed for the eight vehicle categories. The following observations have been made from these comparisons:

- As with tyre wear, only the EMEP method incorporates a speed dependency. The other models use single emission factors for different vehicle categories and size fractions. As noted earlier none of the models define brake wear emissions in sufficient detail to allow for predictions to be made at specific locations, such as in the vicinity of traffic lights, roundabouts or junctions.

- For PM_{10} the RAINS, CEPMEIP and MOBILE 6.2 emission factors differ substantially for all vehicle categories. The range of values in these three models is covered by the variation in the EMEP function according to speeds (again, with a vehicle load factor of 0.5).

- The PM_{10} emission factors in the CEPMEIP database are generally equivalent to the values in the EMEP function at a speed of around 65-75 km/h. It is possible that both are reasonably accurate, in general terms, with CEPMEIP representing average driving conditions.

- MOBILE 6.2 does not provide a $PM_{2.5}$ emission factor for brake wear. For brake wear, $PM_{2.5}$ forms a substantial part of PM_{10}, and $PM_{2.5}$ is a commonly used metric. This is therefore a significant omission.

- For $PM_{2.5}$, the EMEP emission factors bear a closer resemblance to those in RAINS than to those in CEPMEIP, but again the speed effect in the EMEP model is as great as the difference between the emission factors in the other two models.

An example of the effect of the vehicle load factors on PM_{10} emissions, as used in the EMEP method, is given in Figure 6. Vehicle load clearly has a larger absolute effect on emissions at lower average speeds.

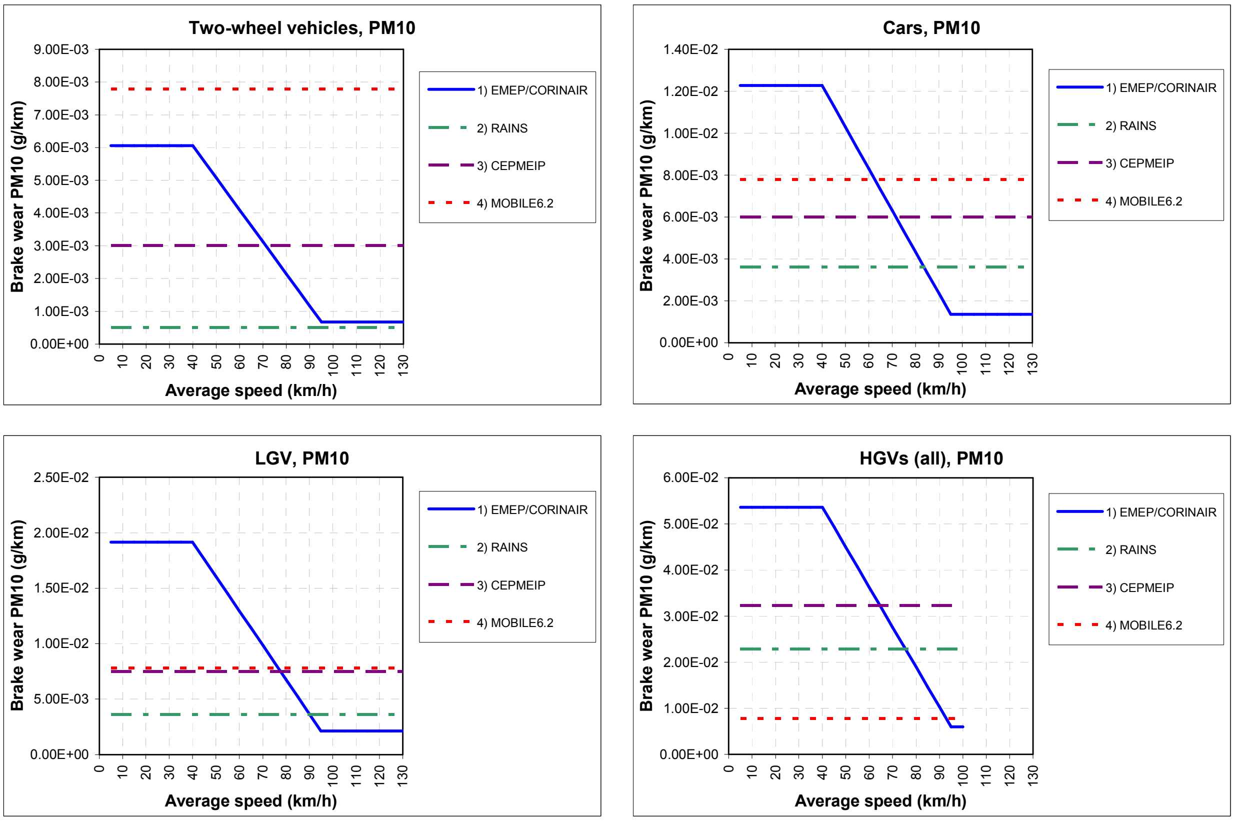

Figure 4: PM_{10} emission factors for brake wear by vehicle type and emission model.

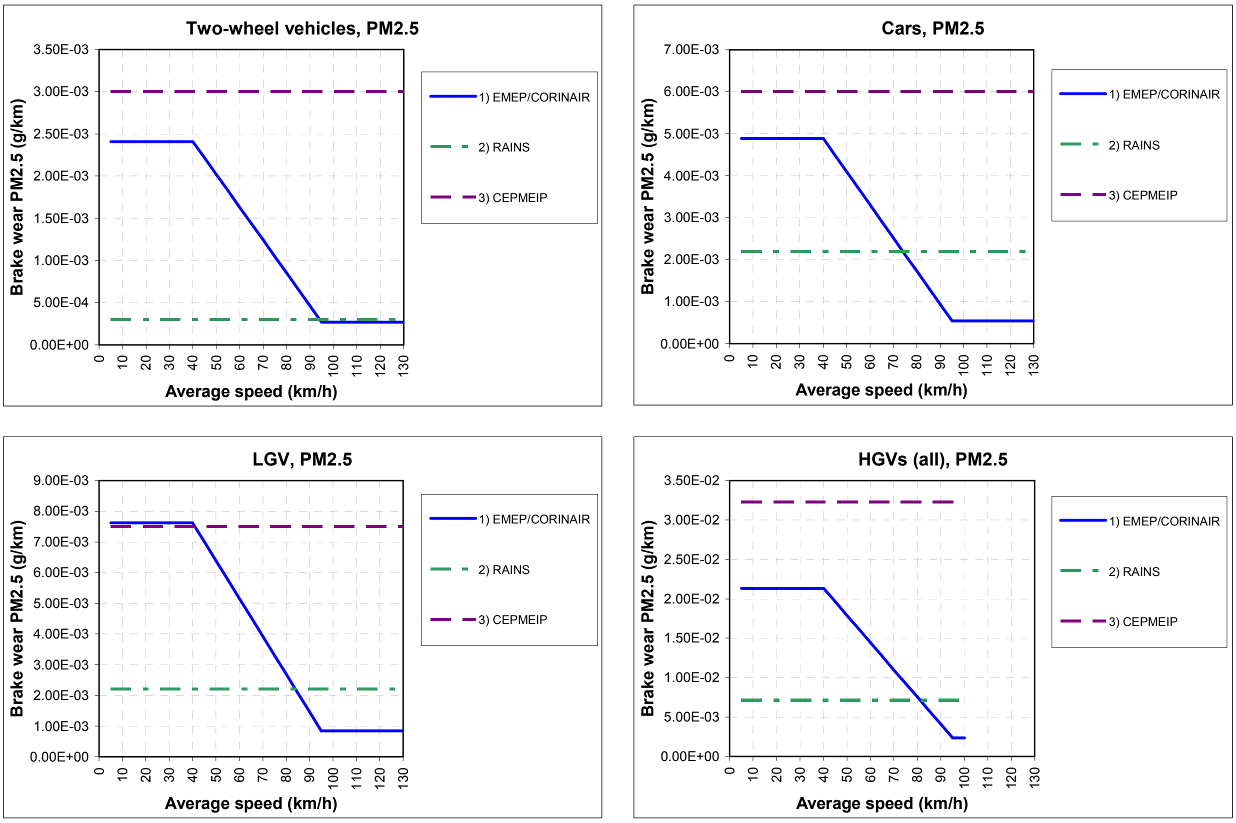

Figure 5: $PM_{2.5}$ emission factors for brake wear by vehicle type and emission model.

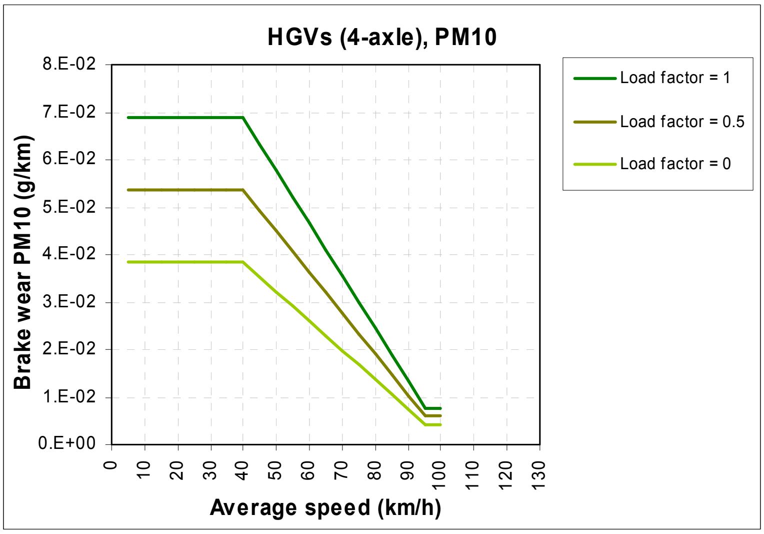

Figure 6: PM_{10} emission factors for brake wear on 4-axle HDVs as a function of vehicle load.

2.3.3 Road surface wear

The emission factors for road surface wear in RAINS and CEPEMIP are shown in Table 3. The EMEP methodology uses the RAINS emission factors, and MOBILE 6.2 does not include road surface wear explicitly as an abrasion source. Furthermore, RAINS does not distinguish between cars and light good vehicles (it only refers to light-duty vehicles), neither RAINS nor CEPMEIP specifies emission factors for different types of HDV, and neither provides emission factors for PM_1 or $PM_{0.1}$.

Table 3: Emission factors for road surface wear.

Vehicle category	Model	Emission factor (mg/km)				
		TSP	PM_{10}	$PM_{2.5}$	PM_1	$PM_{0.1}$
2-wheel vehicles	RAINS	6	3	1.6	-	-
	CEPMEIP	73	3.65	0	-	-
Cars	RAINS	15	7.5	4.2	-	-
	CEPMEIP	145	7.25	0	-	-
LGVs	RAINS	15	7.5	4.2	-	-
	CEPMEIP	190	9.5	0	-	-
All HDVs	RAINS	76	38	21	-	-
	CEPMEIP	738	26.9	0	-	-

For PM_{10} the values in RAINS are broadly similar to those in CEPMEIP. For TSP and $PM_{2.5}$, on the other hand, there are some rather large differences. The TSP emission factors in CEPMEIP are roughly an order of magnitude higher than those in RAINS, and CEPMEIP assumes that no $PM_{2.5}$ is emitted as a result of road surface wear. However, although the sources of information used to derive the CEPMEIP emission factors were known, the precise manner in which the information was used by TNO to develop the model was nor. Hence, it was difficult to determine the reasons for the differences in the predictions.

2.3.4 Summary

For most vehicle categories, the different models/databases produce substantially different emission factors, but in the absence of independent test data it was not possible to assess the absolute accuracy of the different approaches. The EMEP method, which is currently used in the NAEI, is the most detailed approach, incorporating corrections for both speed and, in the case of HDVs, load. For tyre wear, different emission factors are also provided in EMEP for different types of HDV, based on the number of axles. It was therefore concluded that none of the other three models or databases considered in the evaluation would currently offer any advantage over the EMEP method for application in the UK. Consequently, no further consideration was given to development of the RAINS, CEPMEIP and MOBILE 6.2 approaches for modelling abrasion-related emissions.

3 Model development: abrasion sources

The abrasion model development phase of the work proceeded along the following lines:

- Further development of the EMEP method.
- Adaptation of the tyre wear model in HDM-4.
- Use of brake wear data from the US relating to PM emissions per braking event.

However, given that there have been few new measurements and methodological developments for the abrasion sources in recent years, Task 2a(i) was extended to include prediction methods for total non-exhaust particles.

These different approaches and investigations are described in the following Sections.

3.1 Further development of the EMEP method

Other than the information included in the models considered here, little additional source-specific information, either data or methodology, has been published. The review by Boulter (2005a) only identified the new sources of data shown in Table 4. A number of other recent studies were found to have obtained emission factors for total non-exhaust particles, but as these emission factors include resuspension they are not relevant to the EMEP modelling approach.

Table 4: Data not included in existing models.

Data source	Vehicle category	PM source	PM_{10} (mg/km)
Luhana *et al*. (2004)	LDV	Combined tyre & brake wear	6.9
		Road surface wear	3.1
	HDV	Combined tyre & brake wear	49.7
		Road surface wear	29
Kupiainen *et al*. (2005)	LDV	Tyre wear	0.5-0.6[6]
	LDV	Road surface wear	8.5-10.5[7]

The combined tyre and brake wear emission factors reported for the Hatfield Tunnel by Luhana *et al*. (2004) are rather difficult to assess. In fact, the authors noted that traffic conditions in the tunnel were generally fluid, and a possible inference may be that braking was minimal. If it is therefore assumed that the PM_{10} emission factors are largely due to tyre wear, the value of 6.9 mg/vkm for LDVs is in broad agreement with the emission factors for cars in the EMEP method, but for LGVs it is closer to the values in RAINS and CEPMEIP. The combined value for HDVs of 49.7 mg/vkm is close to the maximum tyre wear value for all models and all types of HGV. It was concluded that these combined

[6] Calculation by the authors.
[7] Calculation by the authors.

values could not therefore be used for further model development, on account of the difficulty of separating the tyre wear and brake wear contributions.

The road surface wear emission factors do stand up to direct comparison with the values used in the other models, and were available for use in further mode development. However, the PM_{10} emission factor for LDVs of 3.1 mg/vkm is lower than the values used in RAINS (and EMEP) and CEPMEIP (Table 3), and even lower than the $PM_{2.5}$ emission factor used in RAINS. For HDVs, on the other hand, the value of 29 mg/vkm derived by Luhana *et al.* (2004) compares favourably with the values presented in RAINS and CEPMEIP. However, for road surface wear the EMEP method is based upon the RAINS database. In RAINS, the emission factors for road surface wear are calculated as the difference between total emission factors for non-exhaust PM, and the combined tyre, wear and resuspension emission factors. This method is likely to be prone to error, as the emission factor for resuspension can be large. Furthermore, there appears to be no published information describing the exact method used. Consequently, the results of Luhana *et al*. (2004) for road surface wear could not be included in the EMEP method.

The results of Kupiainen *et al*. (2005) were used by Boulter (2005a) to estimate tyre and road surface PM_{10} emission factors for LDVs. For tyre wear, the resulting values are an order of magnitude lower than the values used in the models considered in the previous Chapter. The road surface wear emission factors are in broad agreement with those in RAINS and CEPMEIP. However, these values were not reported directly by the authors, and therefore need to be viewed with caution. Furthermore, the incorporation of new emission factors for road surface wear in the RAINS method requires co-operation with the model developers (IIASA), and this was considered to be beyond the scope of the project.

3.2 Adaptation of HDM-4 tyre consumption model

Originally developed by the World Bank, the Highway Development and Management Model (HDM) has become widely used as a planning and programming tool for highway authorities. HDM is a computer model which simulates physical and economic conditions over the period of analysis, usually a life-cycle, for a series of alternative strategies and scenarios specified by the user. PIARC has led the management and coordination of international HDM-4 activities since 1998.

A mechanistic tyre consumption model has been proposed for inclusion in HDM-4 (Bennett and Greenwood, 2001), and this is described in Appendix B. The mechanistic model can be used to calculate tyre consumption in dm^3/1000 tyre-km. In order to convert this to mass of tyre material worn per vehicle-km, the following default values were assumed:

- The number of tyres per vehicle : as in Table 2
- The average mass per vehicle : as in Table 2
- The number of wheels per vehicle : as in Table 2
- The density of tyre tread : 0.91 g/cm^3
- Average road radius : 573 m

Figure 7 shows the relationship between tyre wear and vehicle speed for cars, as predicted using the default values provided for the mechanistic model. For comparison, Figure 7 also contains some experimental data on car tyre wear rates from the PARTICULATES project (Luhana *et al*., 2004), and (unreferenced) information taken from the Pirelli web site[8]. The former were used to develop the speed-dependence in the EMEP method. The latter takes the form of tyre lifetime, normalised to a

[8] http://www.pirelli.com/en_42/tyres/about_tyres/tyre_advice/tread_wear_speed.jhtml

speed of 70 km/h. Little supporting information is provided by Pirelli, and so it was assumed that the data relate to car tyres. It was also assumed that a typical car has a total tyre wear factor of 100 mg/vkm.

The mechanistic HMD-4 model predicts that tyre wear increases exponentially with increasing speed when speed variation is low. When there is high speed variation, a minimum tyre wear factor occurs at around 70 km/h. For medium speed variation conditions, the HDM-4 predictions appear to concur broadly with the data presented by Pirelli. The data of Luhana *et al*. (2004), on the other hand, appear to indicate a reduction in tyre wear with speed. A single road radius value of 573 m was used in HDM-4 to generate Figure 7. In fact, the HDM-4 model is rather sensitive to the average road curvature (especially at higher speeds); wear factors for three different values of the road curvature are shown in Figure 8.

In order to convert the total tyre wear to a mass of PM_{10} per vehicle-km, the following assumptions were required:

- 5% of total tyre wear was assumed to be emitted as PM_{10}, based on typical values for tyre wear (~100 mg/vkm) PM_{10} emissions (~5 mg/vkm) for cars in the literature (Boulter, 2005a). This is a rather crude assumption.
- The proportion of tyre wear emitted as PM_{10} was independent of vehicle speed. Again, this is a rather crude assumption. It is possible that, under high-wear conditions, less of the wear material is released as airborne PM, as larger shreds of tyre tread being removed from the tyre. Some early evidence of this was presented by Pierson and Brachaczek (1974).

Plots of PM_{10} emission factors derived using in this way are shown in Figure 9.

For cars, a comparison between the EMEP and HDM-4 PM_{10} predictions is shown in Figure 10. It is clear from this comparison that the two methods give different results. In the case of the EMEP method, tyre wear PM_{10} emissions decrease as speed increases, whereas using the HDM-4 method emissions increase with increasing speed.

The HDM-4 model has been designed to consider the main factors influencing tyre consumption. However, according to Bennett and Greenwood (2001), although the mechanistic theory of tyre consumption is well founded it has not been fully implemented due to an inability to obtain satisfactory model parameters. In its place, a model based on HDM-3, but using adjustment factors has been implemented. This model is not entirely satisfactory either, as the factors adopted will lead to discontinuities in the predictions. Until further work is performed to develop suitable parameters to apply the full theoretical model, the predictions from the HDM-4 model for tyre consumption should be viewed with caution.

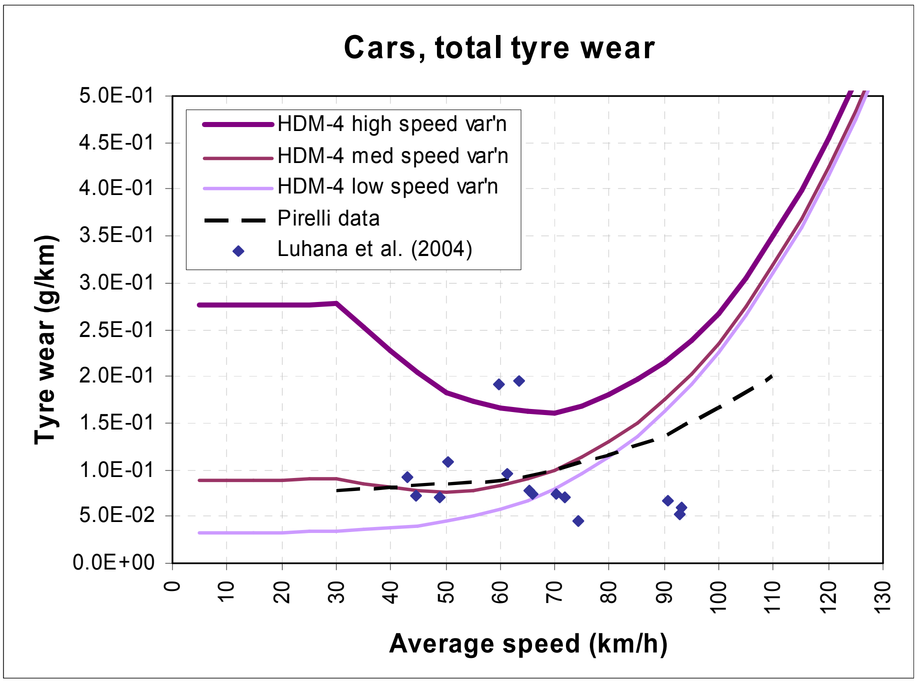

Figure 7: Tyre wear predictions based on mechanistic model proposed for HDM-4 (road radius = 573m). Data from the PARTICULATES project (Luhana *et al*., 2004) and Pirelli are shown for comparison.

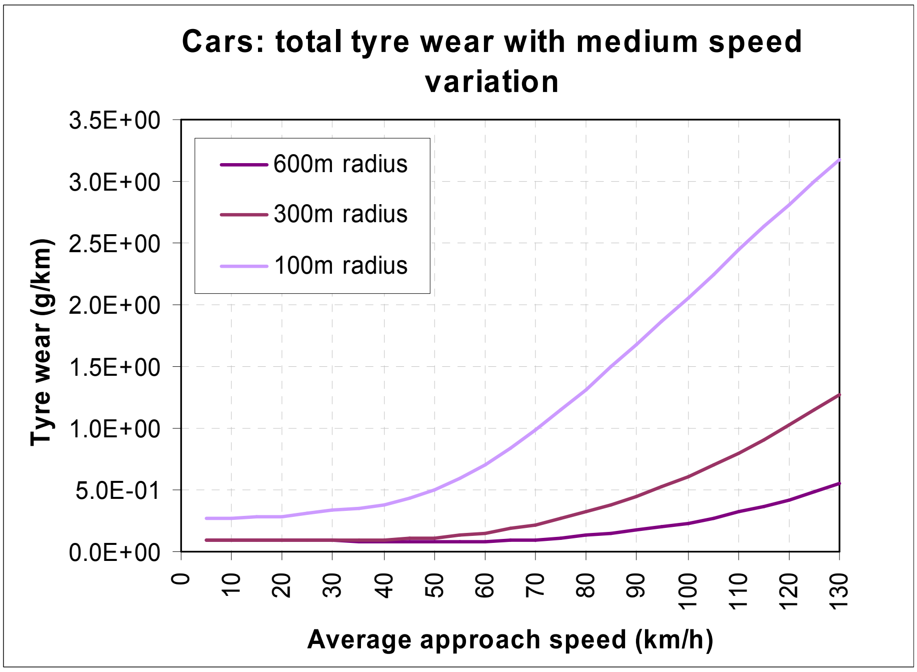

Figure 8: Car tyre wear as a function of average road curvature based on mechanistic model proposed for HDM-4.

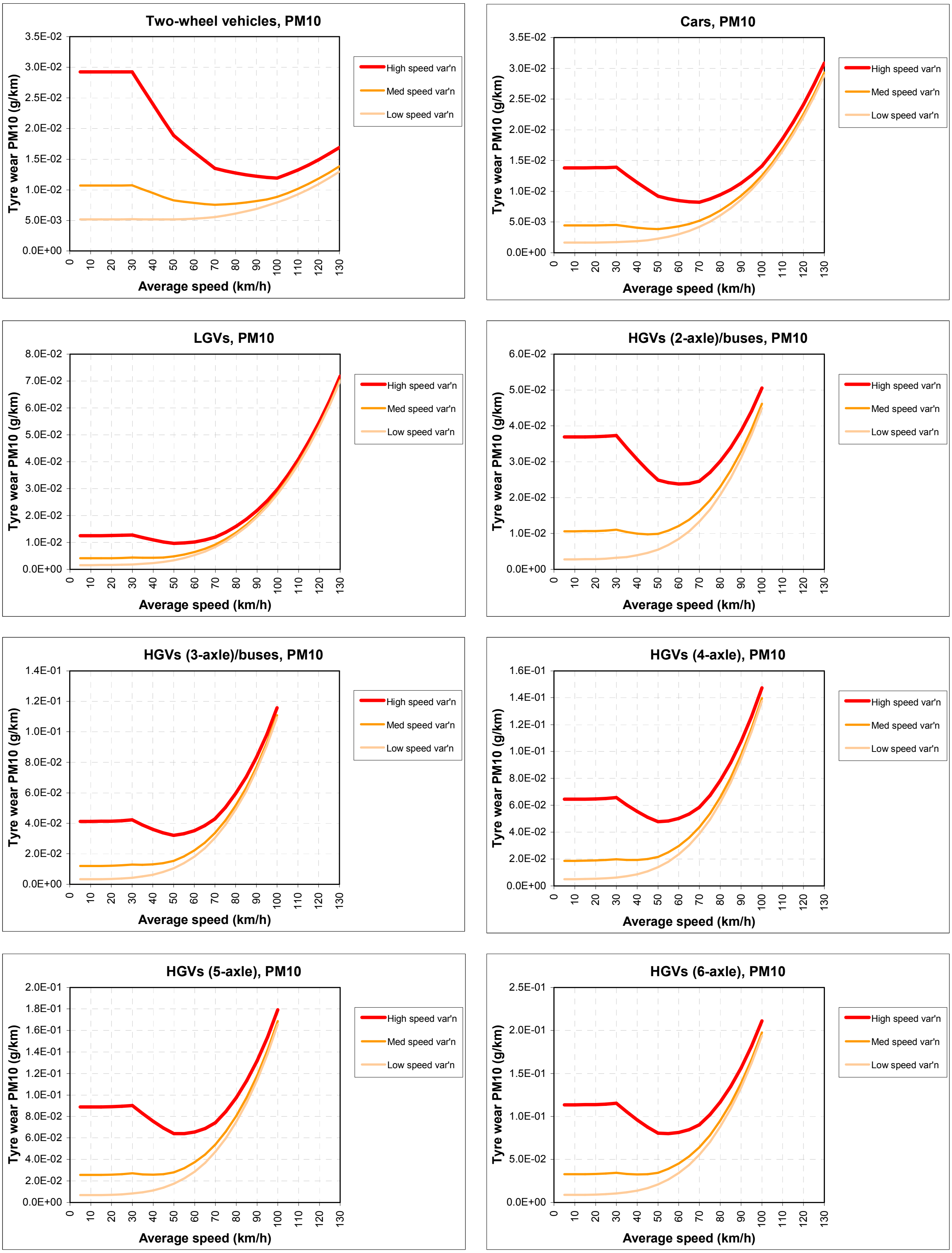

Figure 9: PM$_{10}$ emissions for different vehicle categories, based on HDM-4 tyre consumption model for a road radius of 573m and three levels of speed variation.

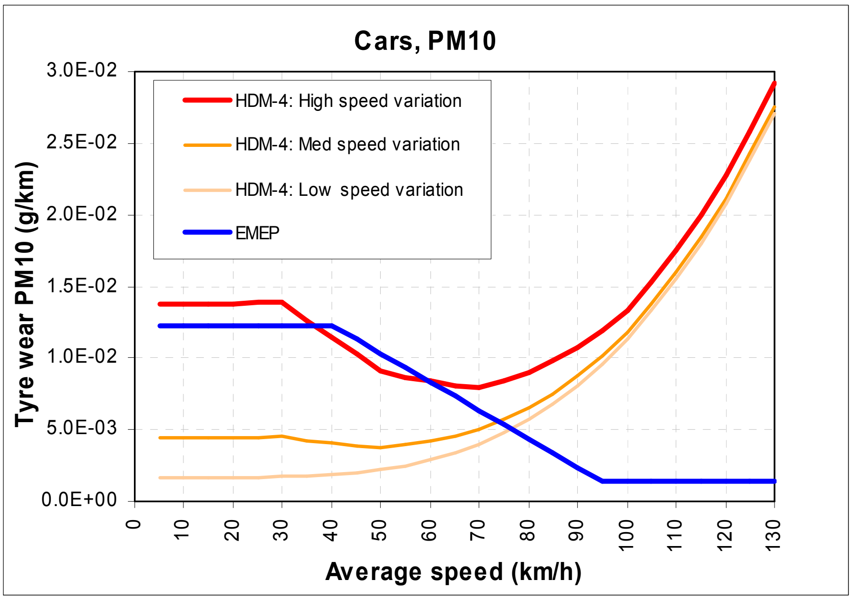

Figure 10: Emissions of PM_{10} due to car tyre wear predicted using the EMEP method and a method based upon the mechanistic tyre consumption model in HDM-4 (for an average road radius of 600 m and three levels of speed variation.

3.3 US data on brake wear emissions per stop

A few recent studies in the United States have yielded limited brake wear PM emission factors in terms of mass per stop, and for stops of differing severity (Garg *et al.*, 2000; Sanders *et al.*, 2003). TRL holds a large database of driving patterns for different traffic situations. The possibility of combining these two types of information to derive new emission factors was investigated.

However, brake wear data currently only exist for a limited number of braking conditions, and therefore the development of a prediction method according to this approach has not yet been possible. It should also be noted that the reported emission factors are measured on a laboratory dynamometer under conditions optimized for collecting the airborne particles. Real-world emissions factors are much more difficult to derive, since they require knowledge of the amount of wear debris becoming attached to the vehicle. Sanders *et al*. (2003) attempted to investigate this using wind tunnel data, and estimated that 50-70 % of the emissions measured on the brake dynamometer would escape the vehicle in real world driving. The uncertainty in this value may be larger than that associated with a lack of completeness of the data set on stopping conditions (Mariq, 2005).

3.4 Models for predicting total non-exhaust PM emissions

Wide ranges of values have been reported in the European literature for the PM_{10} emission factor for total combined non-exhaust sources (Boulter, 2005a):

- Light-duty vehicles 4-92 mg/vkm
- Heavy-duty vehicles 70-1270 mg/vkm
- Motorcycles[9] 23 mg/vkm

[9] One study only.

The review by Boulter (2005a) also identified the following models for predicting total non-exhaust PM emissions:

- USEPA AP-42 model for paved road dust
- Modified AP-42 for use in Germany
- German traffic situation model
- SMHI model
- VLUFT model
- Model from the DAPPLE project

The modified AP-42 for use in Germany was superseded by the German traffic situation model, and is therefore given no further consideration here. The remaining models are described briefly in this Section of the Report.

3.4.1 USEPA AP-42 model for paved road dust

The AP-42 provides methods to calculate 'fugitive dust' emissions from unpaved and paved roads, as described in Appendix D. The fugitive dust methods include all traffic PM sources, including exhaust emissions, abrasion products and resuspension, but have been subjected to a great deal of scrutiny. An incisive critique of the AP-42 method for paved roads by Venkatram (2000) found a number of shortcomings with the method. He concluded that the AP-42 model shows that it is not likely to provide adequate estimates of PM_{10} emissions from paved roads. Because the model has little mechanistic basis, it relies on an input variable, the silt loading, that cannot be measured unambiguously. An analysis of those data used to develop the empirical model showed that different but equally plausible empirical models could be developed by using different subsets of the data set, and that these models provided mean emission factor estimates that could differ by a factor of two. Venkatram argued that the conclusions drawn from the study of the paved road emissions model apply to most empirical models that lack a mechanistic foundation.

The results from independent evaluations of the AP-42 method (Zimmer *et al*., 1992; Kantamaneni *et al*., 1996) have highlighted the problem of basing a model on a particular data set. According to Fitz and Bufalino (2002), the AP-42 states that the silt loading reaches an equilibrium value without the addition of fresh material. If equilibrium is attained, then the emission rate should go to zero, although this is not what the paved road equation predicts. Therefore, it is difficult to understand how this equation could be universally applicable unless the material is continuously replaced, a phenomenon which for most public roads is not likely. Gustafsson (2003) has also identified a number of problems with the approach, including the fact that the amount of silt deposited is much less than the amount emitted due to low deposition speed compared with vehicle-induced turbulence and the wind, the silt loading is not proven to be related to the PM_{10} (or $PM_{2.5}$) concentrations, and the silt loading is hard to generalise. These factors have led to very variable levels of agreement with measurements.

According to APEG (1999), the AP-42 method is based on old measurements near dusty roads. Such conditions are not thought to be relevant to the UK.

3.4.2 German traffic situation model

The USEPA AP-42 model was modified by Rauterberg-Wulff (2000) for use in Berlin. Further modifications, involving the separation of the exhaust and non-exhaust contributions, have been made by Gamez *et al.* (2001). A validation exercise conducted by Lohmeyer *et al*. (2004a) alongside an arterial road in Karlsruhe indicated that the modified method over-predicted roadside non-exhaust PM_{10} concentrations. Over-predictions were also observed at motorway sites in Germany (Lohmeyer *et al*., 2003). By 2004, the modified AP-42 method had been abandoned in favour of allocating emission factors to 'traffic situations' (Lohmeyer *et al*., 2004a) – see Appendix E. This approach was found to give a better performance.

3.4.3 SMHI model

A model was developed by the Swedish Environmental and Health Protection Administration for resuspension, proposed by Johansson *et al*. (1998) (cited in Rauterberg-Wulff, 2000 and Gustafsson, 2003). This model was briefly described by Boulter (2005a), but has since been updated (Omstedt *et al*., 2005). The new version of the model is described in Appendix F, but the full range of input parameters is not yet available for the UK; it appears that a local calibration is required.

3.4.4 VLUFT model

NILU's VLUFT model (Tønnesen, 2000) incorporates a resuspension module. The general form of the model is shown in Appendix G, but there is little supporting documentation in English. Gustafsson (2003) highlighted the potential problems of this approach as being the dependency of resuspension on fine particle emissions, the limited consideration of meteorology, and the limited consideration of differences in the pavement. As with the SMHI model, the full range of input parameters is not yet available for the UK.

3.4.5 Model from the DAPPLE project

Patra *et al*. (2005) designed an experiment to observe and quantify, as far as possible, the dispersion of a bulk sample of dust along a trafficked section of road in an urban environment. The principal aim of the study was to identify and quantify the processes through which particulate matter is lost from the road surface due to traffic. The rate of movement of material along the road in the direction of traffic flow was estimated by observing the difference in arrival time of elevated concentrations of micrometer-size particles in roadside air adjacent to and a short distance downstream of a section of road onto which rock salt was applied, on a three-lane, one-way building-lined street. A model which accounts for the results is briefly summarised in Appendix H.

3.5 Models to be used in Task 2a(ii)

In order to estimate emissions due to resuspension, a number of separate models were carried forward into the next stage of the project - Task 2a(ii). The main models were EMEP, RAINS, and CEPMEIP. The HDM-4 tyre wear model and the German traffic situation model were also tested. The models are summarised in Tables 5 and 6, and the rationale for the selection of these models (and the exclusion of others) is described in the following paragraphs.

Table 5: Models to be used in Task 2a(ii) and predicted metrics.

Model	Tyre wear			Brake wear			Road surface wear			Total non-exhaust		
	PM_{10}	$PM_{2.5}$	PM_1	PM_{10}	$PM_{2.5}$	PM_1	PM_{10}	$PM_{2.5}$	PM_1	PM_{10}	$PM_{2.5}$	PM_1
EMEP	✓	✓	✓	✓	✓	✓	✓	✓	-	-	-	-
RAINS	✓	✓	-	✓	✓	-	✓	✓	-	-	-	-
CEPMEIP	✓	✓	-	✓	✓	-	✓	✓	-	-	-	-
HDM-4	✓	-	-	-	-	-	-	-	-	-	-	-
German TS model	-	-	-	-	-	-	-	-	-	✓	-	-

Table 6: Models to be used in Task 2a(ii): user-defined inputs (all models require the number of vehicles in each category for a given time period).

Model	User-defined input parameters	
	For all traffic	For individual vehicle categories
EMEP		-Average speed (5-130 km/h) -Vehicle load (HDVs only)
RAINS/CEPMEIP	N/A	N/A
HDM-4	-Average road radius	-Average speed (5-130 km/h) -Speed variation (low/med/high) -Estimated vehicle mass (tonnes) -Number of wheels per vehicle -Tyre density (g/cm^3)
German TS model	N/A	N/A

N/A = Not applicable

In Chapter 2 it was concluded that the EMEP method should be used for predicting emissions due to abrasion, as it is the most detailed approach. The existing EMEP method was described by Boulter (2005a), and is currently used in the NAEI. As this model is based upon a wider information base than CEPMEIP and RAINS, and provides the user with some flexibility in terms of vehicle speed and vehicle load. Although it was also concluded that none of the other three abrasion models databases considered in the evaluation would currently offer any advantage over the EMEP method for application in the UK, the RAINS and CEPMEIP databases could also be used to provide an indication of the sensitivity of the abrasive source estimate to the model/database used. Hence, the main emphasis in Task 2a(ii) was placed on these three models. The EMEP method was used to provide 'baseline'(*i.e.* current NAEI) estimates of PM emissions from abrasion sources. In order to estimate PM_{10} emissions due to resuspension, Task 2(ii), DEHRM applied a method based upon simultaneous measurement of PM_{10} and $PM_{2.5}$ in ambient air at roadside and background monitoring sites. This work is described in Chapter 4. During the application of such a method, it is typically assumed that the roadside increment of $PM_{2.5}$ is due to exhaust emissions, and that the coarse particle increment (PM_{10}-$PM_{2.5}$) is due to the combined non-exhaust sources (abrasion and resuspension). However, the EMEP method includes $PM_{2.5}$ emission factors for the abrasion sources, and this will need to be taken into account. The RAINS and CEPMEIP databases were used to provide an indication of the sensitivity of the abrasive source estimate to the model/database used.

The relatively detailed mechanistic tyre consumption model of Bennett and Greenwood (2001) used in HDM-4 (combined with a PM_{10} conversion routine) is pertinent only to the estimation of tyre wear emissions, and would therefore contribute little to an estimate of total abrasive emissions. However, the HDM-4 tyre wear model was also recommended for use in Task 2a(ii) in order to assess the effects of applying a detailed model to calculate tyre wear emissions. Given the reservations expressed by the model developers concerning its validity, this approach was only used for testing purposes, and the results will not be implemented.

Although there may be concerns about the representativeness of the German traffic situation model for the UK, it was also considered appropriate to include this model for further sensitivity testing.

The AP-42 model approach was rejected as being largely discredited. It performs poorly even in the US situation for which it was designed, and requires silt loading data as an input, which are not available for the UK. The VLUFT method is available for use in the UK, but only calibration factors for Oslo and Stockholm are known. For use in the UK, a local calibration would be strongly recommended. Furthermore, a weakness in the model is an assumption made for total non-exhaust $PM_{2.5}$ emissions (taken to be 0.4 x the coarse fraction). The validity of this assumption would also need to be investigated. The VLUFT approach may merit therefore further study, but there was insufficient time to investigate this further in this project. Similarly, it was felt that it would not be able to fully assess the model from the DAPPLE project in the available time frame.

4 Model development: resuspension

4.1 Introduction

In the earlier sections of this report, the models describing the generation of non-exhaust particles through abrasion processes were reviewed. In Task 2a(ii), this information was combined with air pollution and traffic data to estimate the magnitude of quantifiable non-exhaust sources for particular localities, and specifically for Marylebone Road in London, for which excellent air quality and traffic data sets are available. The assembled data are then used to estimate the magnitude of resuspended particulate matter, and to determine emission factors for this source.

The major part of the work was conducted with data from Marylebone Road because of its comprehensive nature and completeness. Data collected during the TRAMAQ programme were examined to test consistency with the findings from the Marylebone Road data analysis.

4.2 Analysis of PM data from Marylebone Road

Daily mean and median values of measured concentrations of PM_{10} ($\mu g/m^3$), $PM_{2.5}$ ($\mu g/m^3$), and NO_x as NO_2 ($\mu g/m^3$), along with traffic data (average speed and fleet composition) calculated from hourly data collected at Marylebone Road between 1 January 2000 and 31 December 2002, have been analysed to determine an estimate for the contribution of resuspension to the overall coarse particle emission from non-exhaust traffic sources. Data from the London Bloomsbury monitoring site were used to represent urban background concentrations. For both the Marylebone Road and Bloomsbury data, values for $PM_{2.5\text{-}10}$ were calculated by difference, and roadside incremental values of PM_{10}, $PM_{2.5}$, $PM_{2.5\text{-}10}$, and NO_x as NO_2 were calculated by subtraction of the concentrations observed at Bloomsbury from those measured at Marylebone Road.

In addition to the primary goal of deriving an estimate of resuspension, the data were examined for evidence of relationships with meteorological factors, principally wind speed, wind direction and precipitation level.

4.2.1 Method for estimating resuspension

The total emissions attributable to non-exhaust sources can be expressed as the sum of the contribution of abrasion sources (brake, tyre, and road wear) and the resuspended component:

$$E_{TOTAL} = E_{TYRE} + E_{BRAKE} + E_{ROAD} + E_{RESUSP} \qquad \text{(Equation 1)}$$

where:

E_{TOTAL}	is the total non-exhaust PM emission
E_{TYRE}	is the PM emission due to tyre wear
E_{BRAKE}	is the PM emission due to brake-wear
E_{ROAD}	is the PM emission due to road surface wear
E_{RESUSP}	corresponds to resuspension emissions

Emissions for tyre, brake and road surface wear can be estimated from emission factors and traffic fleet data. Coupled with an estimate of total particulate matter, Equation 1 can be rearranged to obtain an estimate of resuspension.

The approach taken to estimate the contribution of resuspension relied upon several fundamental assumptions. Firstly, it was assumed that PM_{10} and NO_x behave similarly in the atmosphere, and therefore that the roadside incremental values of the two are closely related. This enables the estimation of the total PM_{10} emissions from the traffic ($\boldsymbol{E_{PM10}}$) on the basis of measured roadside increments of PM_{10} ($\Delta \boldsymbol{PM_{10}}$) and NO_x ($\Delta \boldsymbol{NO_x}$), and calculated NO_x emissions ($\boldsymbol{E_{NOx}}$), according to Equation 2:

$$\boldsymbol{E_{PM_{10}} = E_{NO_x}\left(\frac{\Delta PM_{10}}{\Delta NO_x}\right)} \qquad \text{(Equation 2)}$$

The source strength for NO_x can be estimated from emission factors, the traffic mix, count, and speed data, and is considered to be known with greater confidence than many other traffic-generated air pollutants.

A second assumption in the initial calculations was that the roadside increment $PM_{2.5}$ was solely attributable to vehicular exhaust sources and that non-exhaust emissions were largely confined to the $PM_{2.5\text{-}10}$ size fraction. By apportioning total PM_{10} emissions into $PM_{2.5}$ and $PM_{2.5\text{-}10}$ fractions according to the observed ratios between the concentrations of these size fractions allowed a value for total $PM_{2.5\text{-}10}$ (*i.e.* non-exhaust PM) to be calculated. The assumption is clearly rather crude, especially with respect to the non-exhaust emissions, and a sensitivity calculation is conducted assuming a dividing point at 1 μm diameter.

4.2.2 NO_x emission calculation

The calculation of NO_x emissions was performed using vehicle fleet composition and average vehicle speed data from Marylebone Road, along with NO_x emission factors for different vehicle classes and European emission standards for engine technologies.

The Marylebone Road traffic data were segregated into several vehicle categories, with hourly data on the number of vehicles within each class passing the monitoring site. The vehicle categories incorporated were:

- Two-wheeled motor vehicles (TWMV)
- Cars
- Cars with trailers
- Rigid HGVs
- Articulated HGVs
- Buses

In the NAEI method for calculating NO_x emissions, a further class, light-goods vehicles (LGVs), is included. The number of LGVs at Marylebone Road was estimated by summating the number of cars and cars with trailers, then apportioning this number according to the percentages given in the NAEI tables[10] on national vehicle fleet composition for the split of light-duty vehicles between cars (85%) and LGVs (15%).

NO_x emission factors (g/km) were calculated by applying the polynomial equations given in the NAEI[11] for a range of vehicle speeds from 25 to 60 km/h. Equations are given for various engine technologies and capacities, and different fuel types, and so initial calculations were made for an individual vehicle within each of these categories. Correction factors were then applied to the emission factors to obtain figures considered representative of the situation at Marylebone Road for the years of interest (2000-2002). The first correction was for the fraction of vehicle kilometres travelled by vehicles with a given engine technology. For each engine technology, the respective

[10] http://www.naei.org.uk/other/uk_fleet_composition_projections_v2.xls
[11] http://www.naei.org.uk/datachunk.php?f_datachunk_id=8

emission factors were then corrected for engine size, based on the national fleet statistics obtained from the NAEI. This second step applies only for the case of petrol and diesel motor cars.

A final correction to account for improved fuel technologies was carried out, yielding weighted emission factors for each engine technology and capacity within the individual classes; the sum of these emission factors within each class therefore provided an estimate of the emission factor for a typical fleet-weighted vehicle in that class at a given speed. Based on national statistics of the percentage split of cars into petrol- and diesel-engine vehicles, a composite emission factor for cars was obtained. A similar calculation was applied to petrol and diesel LGVs.

Having calculated fleet-weighted emission factors for each vehicle class for a range of vehicle speeds likely to be encountered in urban areas (25-60 km/h), plots were constructed of vehicle speed against the composite emission factors for each vehicle class. Polynomial trend lines were fitted to obtain the relationship between vehicle speed and NO_x emission factor. Figure 11 shows the plot of vehicle speed (km/h) versus emission factor (g/km) for Marylebone Road for 2001.

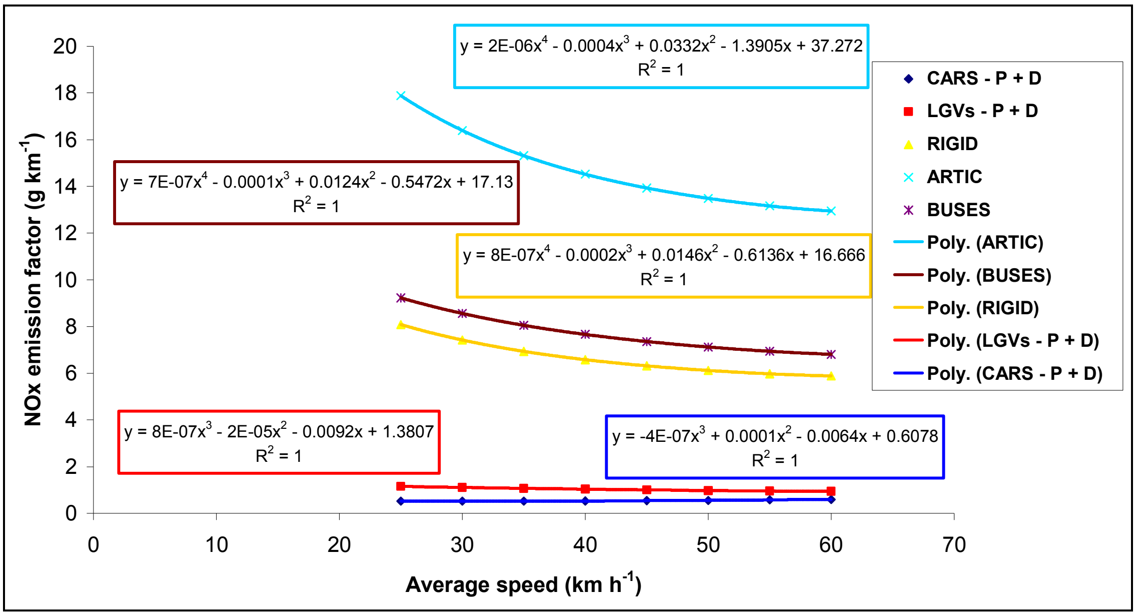

Figure 11: Variation of NO_x emission factors at Marylebone Road (year=2001) with vehicle speed and vehicle category.

The polynomial equations derived from the plots were used along with measured vehicle speed data and disaggregated traffic volume data from Marylebone Road to obtain speed-corrected NO_x emission factors for the traffic fleet at Marylebone Road. Since the emission factors are already corrected for the fraction of vehicle kilometres travelled based on national fleet data, the sum of the emission factors for each vehicle class at Marylebone Road weighted according to the fraction of that vehicle class multiplied by the total number of vehicles in passing the site per hour or per day gives an estimate of total NO_x emissions (g/km.hour or g/km.day).

4.2.3 Particulate matter emissions from abrasion sources

There are a number of sources of airborne particulate matter attributable to abrasion processes. The principal mechanisms are tyre wear, brake-wear, and road surface wear. From the traffic data for Marylebone Road, three methods were used to obtain daily values (g/km.day) for the abrasion emissions.

Two simple methods were employed, based upon the CEPMEIP and RAINS databases. These databases provide emission factors (mg/vkm) for tyre, brake, and road surface wear for various vehicle categories. Appropriate emission factors were selected corresponding to particle emissions in the $PM_{2.5\text{-}10}$ size range. A third, more complex method was also used: the EMEP method. This approach takes into account vehicle speed, and load corrections for heavy goods vehicles. It also provides correction factors to determine the particle emissions in different size fractions. CEPMEIP, RAINS and EMEP all give mass fractions/emission factors for emissions in the PM_{10} and $PM_{2.5}$ size fractions, and EMEP provides additional mass fractions for PM_1 and $PM_{0.1}$. This additional information in the EMEP approach was employed to derive emissions for the $PM_{1\text{-}10}$ size range, as well as the $PM_{2.5\text{-}10}$ fraction.

4.2.4 Estimation of total PM_{10} emissions

Equation 2 illustrates that the estimation of total PM_{10} emissions ($\boldsymbol{E_{PM10}}$, g/km.day) attributable to traffic is possible if data are available on roadside incremental PM_{10} ($\Delta \boldsymbol{PM_{10}}$), roadside incremental NO_x ($\Delta \boldsymbol{NO_x}$), and NO_x traffic emissions ($\boldsymbol{E_{NOx}}$).

Roadside increment PM_{10} and roadside increment NO_x data were plotted against each other, and linear regressions were performed. In order to obtain the best relationship between $\Delta \boldsymbol{PM_{10}}$ and $\Delta \boldsymbol{NO_x}$, days with missing or negative $\Delta \boldsymbol{PM_{10}}$ values were omitted, as were days with missing NO_x data. Furthermore, on certain days the observed $\Delta \boldsymbol{PM_{10}}:\Delta \boldsymbol{NO_x}$ ratio was anomalously high, indicating a source of particulate matter other than road traffic emissions. Consequently, days with $\Delta \boldsymbol{PM_{10}}:\Delta \boldsymbol{NO_x}$ ratio larger than 0.1 were removed from the analysis. The remaining data were used to construct plots of $\Delta \boldsymbol{PM_{10}}$ versus $\Delta \boldsymbol{NO_x}$ for 2000, 2001, and 2002. Ratios of $\Delta \boldsymbol{PM_{10}}:\Delta \boldsymbol{NO_x}$ were obtained from the relationship provided by the linear regressions. Total NO_x emission values for the corresponding year were then multiplied by this ratio to estimate the total PM_{10} emissions.

A factor of 0.4, based on values quoted in the literature (*e.g.* AQEG, 2005; Harrison *et al.,* 2001; Charron and Harrison, 2005) for the $PM_{2.5\text{-}10}:PM_{10}$ ratio was applied to the calculated total PM_{10} emissions to determine total $PM_{2.5\text{-}10}$ emissions at Marylebone Road. For the purpose of deriving mean daily resuspension values, $PM_{2.5\text{-}10}$ emissions calculated using this fixed ratio were considered adequate. Values of $PM_{2.5\text{-}10}$ emissions were also calculated, based on the measured ratio of $PM_{2.5\text{-}10}$: PM_{10}, since the ratio of coarse to fine particles is observed to vary with other influencing factors such as wind speed (*e.g.* Charron and Harrison, 2005) and direction (Harrison *et al.,* 2004). Resuspension estimates calculated with observed $PM_{2.5\text{-}10}:PM_{10}$ ratios were used in the analysis of variation in resuspension with wind speed and direction.

Similar procedures were carried out using a factor of 0.5 to determine values for total emissions in the $PM_{1\text{-}10}$ size fraction. Ayers *et al.* (2004) reported a value for the $PM_1:PM_{10}$ ratio of 0.4, corresponding to 60% of airborne PM_{10} lying in the $PM_{1\text{-}10}$ size range. Gomiscek *et al.* (2004) observed $PM_{1\text{-}10}:PM_{10}$ ratios of 40 – 50% during a study conducted at three urban sites and one rural site in Austria. In Taipei, Li and Lin (2002) documented a mean $PM_1:PM_{10}$ ratio of 0.59, implying that 41% of PM_{10} emissions are comprise of particles in the $PM_{1\text{-}10}$ size range.

4.2.5 Resuspension estimates

Having derived estimates for the total non-exhaust particle emission and emissions arising from abrasion sources, Equation 1 was rearranged to enable an estimation of the resuspended component. $PM_{2.5\text{-}10}$ abrasion emissions (due to tyre, brake, and road surface wear) were subtracted from the total $PM_{2.5\text{-}10}$ emissions to derive a value for resuspension. A second calculation of resuspension was performed excluding road surface wear from the abrasion sources. This is because emissions due to road surface wear are considered highly uncertain, and there is a recognised difficulty in the separation of resuspension and road surface wear components.

4.2.6 Results

Relationships between roadside increment PM_{10} and NO_x and trends in NO_x emissions

The results of the PM_{10} versus NO_x regression analysis are summarised in Table 7. Trend lines fitted using the least-squares method indicate strong correlations between daily mean ΔPM_{10} and daily mean $\Delta \boldsymbol{NO_x}$ for 2000 and 2001, with gradients of 0.0616 and 0.0618 respectively. The small intercepts indicate that NO_x and PM_{10} share the same common source - road traffic emissions - and that road traffic emissions are primarily responsible for the observed roadside increments (Figures 12 and 13). The 2002 data (Figure 14) reveals a poorer relationship between NO_x and PM_{10}. Linear regression indicates a gradient of around 0.07 and an intercept of around -2.5, values which are somewhat larger than those of 2000 and 2001. The R^2 value of 0.60 reflects the greater scatter of the 2002 data compared with the former two years.

Table 7: Summary of results of roadside increment PM_{10} versus NO_x regression analysis.

Year	Intercept	Slope	R2	Slope*
2000	0.8521	0.0616	0.91	0.0637
2001	-0.122	0.0618	0.89	0.0615
2002	-2.4559	0.0659	0.60	0.0585

* With regression line forced through the origin.

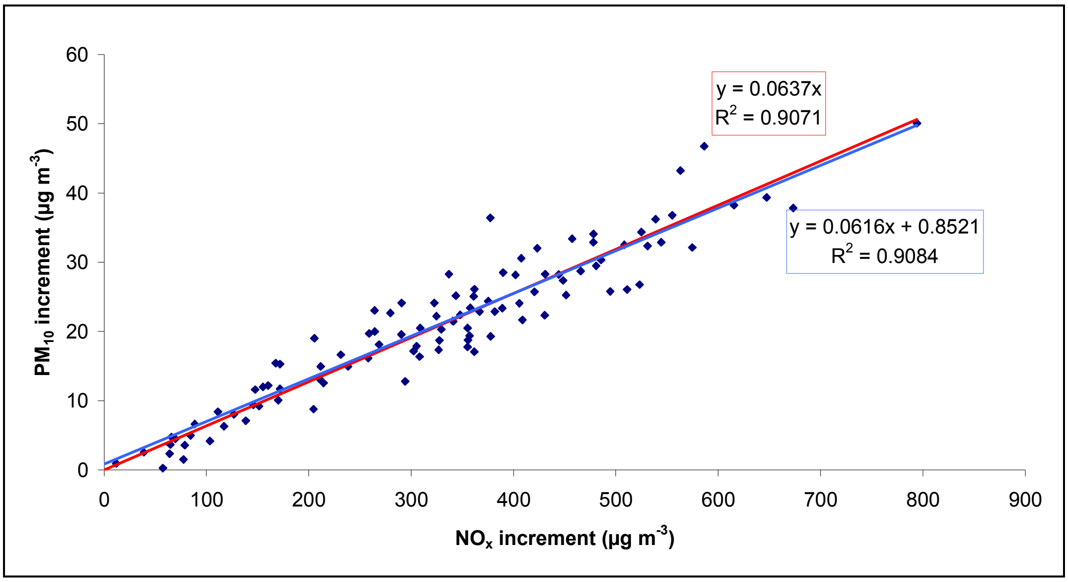

Figure 12: Relationship between roadside increment PM_{10} and NO_x at Marylebone Road (year=2000) using least squares regression. Trend line also shown for intercept constrained to zero.

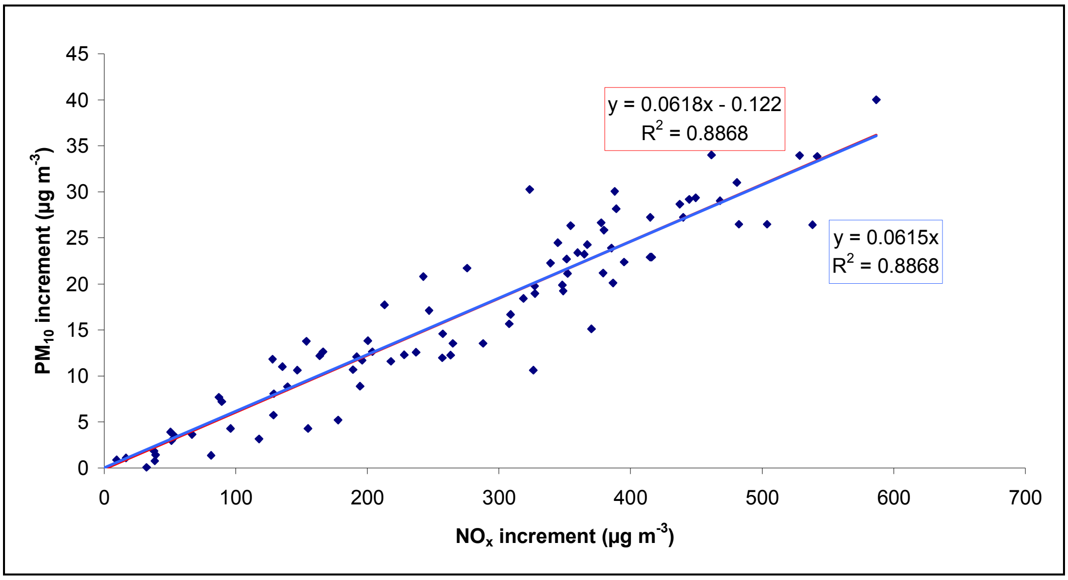

Figure 13: Relationship between roadside increment PM_{10} and NO_x at Marylebone Road (year=2001) using least squares regression. Trend line also shown for intercept constrained to zero.

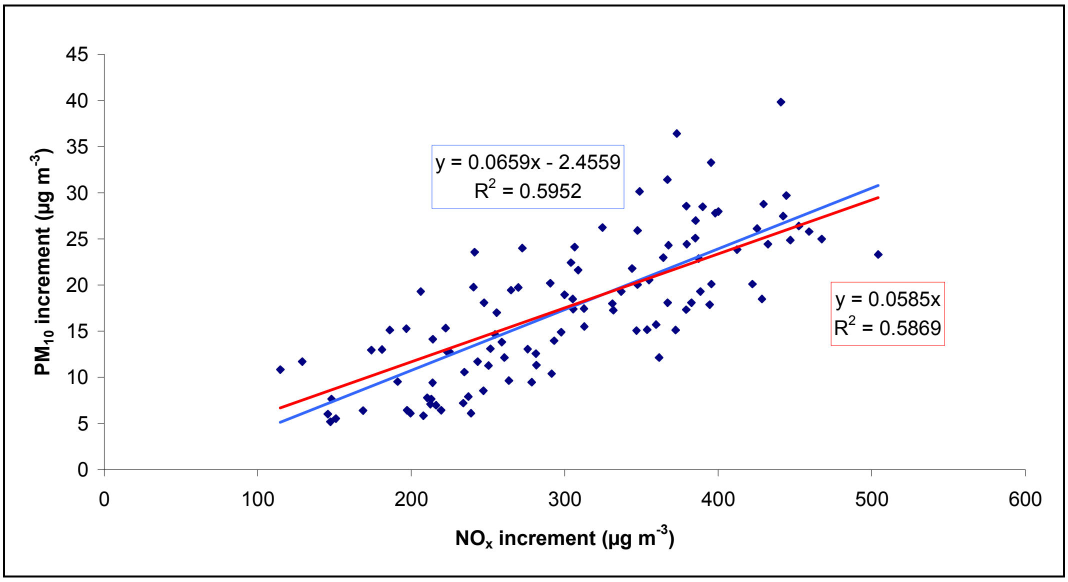

Figure 14: Relationship between roadside increment PM_{10} and NO_x at Marylebone Road (year=2002) using least squares regression. Trend line also shown for intercept constrained to zero.

Tabulated data summarising fleet-weighted emission factors and NO_x emissions for Marylebone Road are shown in Table 8. It is apparent from this data that cars are responsible for 29.3% to 34.5% of all NO_x emissions at Marylebone Road. The rigid HGV category accounts for a slightly higher percentage of NO_x emissions (37.2 % to 39.8 %), and is the category which provides the largest contribution. Mean values of the total estimated fleet-weighted emission factor for all vehicles range from 1.25 g/km to 1.44 g/km. In a recent study, Kohler *et al.* (2005) determined similar values for NO_x emissions (1.08 g/km) from vehicles travelling along a motorway in Germany, although the closeness of the estimates is probably fortuitous given the differences in road type and other factors.

Table 8: Summarised fleet-weighted NO_x emission factors and total NO_x emissions at Marylebone Road.

Year and statistic	Fleet-weighted NOx emission factor (g/vkm)						Total emissions (g/km.day)
	Car	LGV	Rigid HGV	Articulated HGV	Bus	All vehicles	
2000							
10th %ile	0.477	0.103	0.217	0.077	0.092	1.052	83,011
90th %ile	0.535	0.109	0.693	0.292	0.116	1.670	147,759
Arithmetic mean	0.498	0.105	0.537	0.200	0.104	1.444	121,804
2001							
10th %ile	0.406	0.099	0.225	0.080	0.087	0.970	66,581
90th %ile	0.454	0.105	0.697	0.293	0.114	1.595	133,586
Arithmetic mean	0.424	0.101	0.541	0.198	0.100	1.364	105,013
2002							
10th %ile	0.351	0.093	0.212	0.086	0.089	0.909	64,357
90th %ile	0.388	0.099	0.655	0.261	0.126	1.465	111,806
Arithmetic mean	0.365	0.095	0.496	0.184	0.107	1.247	92,721

Assessing the inter-annual trend in total NO_x emissions, it is clear that improvements in vehicle engine technologies and their greater penetration into the vehicle fleet, coupled with fuel technology advances, have resulted in an overall reduction in emissions. Total emissions for 2000 are estimated to be more than 121 kg/km.day, compared with around 92 kg/km.day for 2002 - a reduction of 24%. Examination of the airborne NO_x concentrations at Marylebone Road (Table 9) reveals that the annual mean of daily concentrations has reduced by 26% between 2000-2002, closely reflecting the large decreases in NO_x emissions. The 2002 annual mean concentration is 300 $\mu g/m^3$, compared with 404 $\mu g/m^3$ for 2000.

Table 9: Summary of daily mean particulate matter and NO_x data from Marylebone Road.

Year and statistic	PM_{10} grav. $\mu g/m^3$	$PM_{2.5}$ grav. $\mu g/m^3$	$PM_{2.5-10}$ grav. $\mu g/m^3$	NO_x as NO_2 $\mu g/m^3$
2000				
10th %ile	26.6	17.7	8.9	156
90th %ile	66.0	46.7	23.8	628
Arithmetic mean	47.8	32.6	15.7	404
2001				
10th %ile	22.3	15.9	6.4	111
90th %ile	61.6	46.7	19.2	547
Arithmetic mean	42.7	30.8	12.8	332
2002				
10th %ile	26.9	16.4	9.9	119
90th %ile	62.0	39.1	26.9	470
Arithmetic mean	44.7	27.8	17.5	300

Total PM_{10}, $PM_{2.5-10}$ and PM_{1-10} emissions

Estimated total PM_{10} emissions data, calculated according to Equation 2 and using NO_x emissions and ratios obtained from the regressions, are shown in Table 10. Also shown are the estimates of total emissions for $PM_{2.5-10}$ and PM_{1-10}, calculated assuming constant factors of 0.4 and 0.5 respectively. Upon first inspection of Table 10 it appears the use of fixed ratios are sufficient for the estimation of the plausible daily mean values of resuspension. It is recognised that over shorter time periods observed ratios of $PM_{2.5-10}$:PM_{10} can vary significantly under the influence of meteorological parameters and traffic intensities, and in these circumstances a ratio based on observed airborne concentrations should be applied.

Table 10: Basic statistics for total PM_{10}, $PM_{2.5-10}$, and PM_{1-10} emissions at Marylebone Road, 2000 – 2002.

Statistic	Total PM_{10} emissions (g/km.day)		
	2000	2001	2002
10th %ile	5,172.2	4,134.0	4,108.5
90th %ile	9,097.1	8,255.1	7,355.6
Arithmetic mean	7,498.1	6,428.2	5,981.9
Statistic	Total $PM_{2.5-10}$ emissions (g/km.day)		
	2000	2001	2002
10th %ile	2,068.9	1,653.6	1,643.4
90th %ile	3,638.8	3,302.0	2,924.2
Arithmetic mean	2,999.3	2,571.3	2,393.8
Statistic	Total PM_{1-10} emissions (g/km.day)		
	2000	2001	2002
10th %ile	2,586.1	2,067.0	2,054.3
90th %ile	4,548.5	4,127.5	3,677.8
Arithmetic mean	3,749.1	3,214.1	2,991.0

Further examination of Table 10 suggests a decrease in total PM_{10} emissions from 2000 to 2002. Data presented in Table 9 indicated that annual mean PM_{10} concentrations have declined by 6% between 2000-2002. The remainder of the apparent reduction in emissions is likely to be the result of the decrease in calculated NO_x emissions resulting from improvements in engine technology affecting the calculated PM_{10} emissions. Due to the use of constant ratios, these trends are duplicated in the $PM_{2.5-10}$ and PM_{1-10} emissions. This is discussed further in a later section.

An analysis of annual-mean daily $PM_{2.5}$ concentrations (Table 9) reveals a decrease in the measured concentrations of approximately 15% between 2000 and 2002. This trend is anticipated for $PM_{2.5}$, which arises mainly from exhaust emissions. In contrast, $PM_{2.5-10}$ concentrations show a slight increase from 2000 to 2002, and as a percentage of PM_{10}, $PM_{2.5-10}$ increase from 32% to 39%. This is consistent with $PM_{2.5-10}$ arising predominantly from non-exhaust emissions, which are not responsive to changing engine technologies. There may, however, be influences from non-traffic sources (*e.g.* construction) which are not readily quantified.

Abrasion source emissions

Basic statistics on emissions due to abrasion processes are displayed in Tables 11 to 18. Abrasion emissions in the $PM_{2.5-10}$ size fraction are shown in Tables 11 to 14, with the PM_{1-10} data in Tables 15 to 18. It is apparent that the three approaches used for the calculation of abrasion emissions differ in terms of which source is considered to be the most important. CEPMEIP attributes the greatest emissions to road surface wear, whilst RAINS considers tyre wear to be the largest contributor to the abrasion emissions. The EMEP method considers brake-wear to be the major source of abrasion-generated particulate emissions. CEPMEIP results in the lowest value for total abrasion emissions, largely due to the inherent assumption that brake-wear particles are confined entirely to the $PM_{2.5}$ fraction. EMEP and RAINS result in similar estimates of total abrasion emissions for Marylebone Road. As a percentage of total $PM_{2.5-10}$ emissions from all abrasion sources, the EMEP approach contributes around 17.5% of abrasion emissions to tyre wear, 26.7% to road surface wear, and 55.8% to brake-wear emissions. Closer examination of the data indicates a decrease in abrasion-related emissions from 2000 to 2002. The emission factors used in the calculations are identical for each year, and so the decrease must relate to a change in traffic mix, traffic volume, or driving pattern (*e.g.* vehicle speed, braking events).

In the case of PM_{1-10} emissions, total emissions are 60-65% higher than the $PM_{2.5-10}$ estimates. Road surface wear values are assumed unchanged from the $PM_{2.5-10}$ estimates since none of the databases provide mass fractions for road wear PM_1. EMEP gives mass factors for PM_1 for both tyre and brake wear emissions. Since CEPMEIP and RAINS do not provide mass fractions for PM_1, the PM_1 mass fractions used in EMEP were applied. As with the $PM_{2.5-10}$ emissions, PM_{1-10} emissions calculated using CEPMEIP are dominated by road wear emissions. Tyre wear is the largest abrasion emission derived using RAINS, and brake-wear is again the greatest when EMEP is used.

Table 11: Summary of tyre wear emissions ($PM_{2.5-10}$) calculated for Marylebone Road.

Year and statistic	$PM_{2.5-10}$ emissions (g/km.day)		
	CEPMEIP	EMEP	RAINS
2000			
10^{th} %ile	345.1	234.8	630.3
90^{th} %ile	494.7	303.1	946.2
Arithmetic mean	432.1	271.8	817.5
2001			
10^{th} %ile	281.5	188.4	518.1
90^{th} %ile	486.0	296.5	929.8
Arithmetic mean	401.4	251.1	760.6
2002			
10^{th} %ile	311.4	214.6	578.3
90^{th} %ile	434.7	265.6	832.8
Arithmetic mean	381.5	241.8	723.2

Table 12: Summary of brake wear emissions ($PM_{2.5-10}$) calculated for Marylebone Road.

Year and statistic	$PM_{2.5-10}$ emissions (g/km.day)		
	CEPMEIP	EMEP	RAINS
2000			
10th %ile	0.0	695.1	170.1
90th %ile	0.0	990.5	289.3
Arithmetic mean	0.0	862.2	242.6
2001			
10th %ile	0.0	545.7	143.0
90th %ile	0.0	972.1	283.5
Arithmetic mean	0.0	799.8	226.5
2002			
10th %ile	0.0	642.5	155.8
90th %ile	0.0	870.3	256.3
Arithmetic mean	0.0	771.9	215.6

Table 13: Summary of road surface wear emissions ($PM_{2.5-10}$) calculated for Marylebone Road.

Year and statistic	$PM_{2.5-10}$ emissions (g/km.day)		
	CEPMEIP	EMEP	RAINS
2000			
10th %ile	672.4	331.8	318.6
90th %ile	901.5	473.0	455.6
Arithmetic mean	802.6	413.6	398.2
2001			
10th %ile	550.5	270.1	259.7
90th %ile	884.6	464.4	447.6
Arithmetic mean	743.9	384.1	369.9
2002			
10th %ile	599.8	305.4	287.1
90th %ile	792.6	415.3	400.2
Arithmetic mean	706.5	369.3	351.5

Table 14: Summary of total abrasion source emission ($PM_{2.5-10}$) data for Marylebone Road.

Year and statistic	$PM_{2.5-10}$ emissions (g/km.day)		
	CEPMEIP	EMEP	RAINS
2000			
10^{th} %ile	1,019.6	1,260.3	1,117.4
90^{th} %ile	1,397.0	1,766.9	1,690.8
Arithmetic mean	1,234.7	1,547.6	1,458.3
2001			
10^{th} %ile	830.6	1,014.8	920.2
90^{th} %ile	1,371.1	1,732.1	1,660.9
Arithmetic mean	1,145.4	1,435.0	1,356.9
2002			
10^{th} %ile	906.1	1,140.8	1,026.3
90^{th} %ile	1,226.2	1,550.6	1,489.2
Arithmetic mean	1,088.0	1,383.0	1,290.3

Table 15: Summary of tyre wear emissions (PM_{1-10}) calculated for Marylebone Road.

Year and statistic	PM_{1-10} emissions (g/km.day)		
	CEPMEIP	EMEP	RAINS
2000			
10^{th} %ile	345.1	704.4	630.3
90^{th} %ile	494.7	909.2	946.2
Arithmetic mean	432.1	815.3	817.5
2001			
10^{th} %ile	281.5	565.1	518.1
90^{th} %ile	486.0	889.6	929.8
Arithmetic mean	401.4	753.3	760.6
2002			
10^{th} %ile	311.4	643.7	578.3
90^{th} %ile	434.7	796.9	832.8
Arithmetic mean	381.5	725.4	723.2

Table 16: Summary of brake wear emissions (PM_{1-10}) calculated for Marylebone Road.

Year and statistic	PM_{1-10} emissions (g/km.day)		
	CEPMEIP	EMEP	RAINS
2000			
10th %ile	526.7	1,036.8	391.2
90th %ile	754.8	1,477.4	583.5
Arithmetic mean	659.3	1,285.9	504.9
2001			
10th %ile	429.6	813.9	321.3
90th %ile	741.5	1,450.0	572.9
Arithmetic mean	612.5	1,192.9	469.3
2002			
10th %ile	475.2	958.3	358.3
90th %ile	663.2	1,298.0	512.8
Arithmetic mean	582.2	1,151.4	446.3

Table 17: Summary of road surface wear emissions (PM_{1-10}) calculated for Marylebone Road.

Year and statistic	PM_{1-10} emissions (g/km.day)		
	CEPMEIP	EMEP	RAINS
2000			
10th %ile	672.4	331.8	318.6
90th %ile	901.5	473.0	455.6
Arithmetic mean	802.6	413.6	398.2
2001			
10th %ile	550.5	270.1	259.7
90th %ile	884.6	464.4	447.6
Arithmetic mean	743.9	384.1	369.9
2002			
10th %ile	599.8	305.4	287.1
90th %ile	792.6	415.3	400.2
Arithmetic mean	706.5	369.3	351.5

Table 18: Summary of total abrasion source emission (PM_{1-10}) data for Marylebone Road.

Year and statistic	PM_{1-10} emissions (g/km.day)		
	CEPMEIP	EMEP	RAINS
2000			
10^{th} %ile	1,545.1	2,072.6	1,339.8
90^{th} %ile	2,151.8	2,859.0	1,985.7
Arithmetic mean	1,894.0	2,514.9	1,720.6
2001			
10^{th} %ile	1,256.9	1,658.2	1,098.5
90^{th} %ile	2,111.8	2,802.2	1,951.3
Arithmetic mean	1,757.8	2,330.3	1,599.8
2002			
10^{th} %ile	1,384.4	1,913.3	1,226.4
90^{th} %ile	1,889.9	2,510.7	1,746.0
Arithmetic mean	1,670.2	2,246.1	1,521.0

Using traffic data and abrasion emissions derived using EMEP, fleet-weighted emission factors for abrasion processes have been calculated and are shown in Tables 19 to 21. The data indicate mean emission factors due to all vehicles of 10.2 mg/km for brake wear, 3.2 mg/km for tyre wear, and 4.9 mg/km for road surface wear. Combining the fleet-weighted emission factors for TWMVs, cars, and LGVs, and buses/rigid HGVs with articulated HGVs, emission factors for LDV and HDV fractions could be determined. Brake wear emission factors were found to be 6.8 mg/km for LDVs, and 3.4 mg/km for HDVs. For tyre wear the corresponding values are 2.5 mg/km and 0.7 mg/km for LDVs and HDVs respectively. Emission factors for road wear were calculated to be 3.0 mg/km for LDVs and 1.8 mg/km for HDVs. Since EMEP provides emission factors for a greater range of vehicle categories, and includes corrections for vehicle speed and heavy goods vehicle load, the use of the abrasion emissions derived using this method were favoured in the calculation of resuspension.

Table 19: Fleet-weighted $PM_{2.5-10}$ tyre wear emission factors for the vehicle fleet at Marylebone Road.

	Fleet-weighted $PM_{2.5-10}$ emission factor (mg/km)					
	TWMV	Car	LGV	Rigid HGV/bus	Articulated HGV	All vehicles
2000						
10^{th} %ile	0.015	2.0	0.4	0.2	0.1	2.7
90^{th} %ile	0.022	2.2	0.4	0.6	0.3	3.6
Arithmetic mean	0.019	2.1	0.4	0.5	0.2	3.2
2001						
10^{th} %ile	0.016	2.0	0.4	0.3	0.1	2.7
90^{th} %ile	0.028	2.2	0.4	0.6	0.3	3.6
Arithmetic mean	0.022	2.1	0.4	0.5	0.2	3.2
2002						
10^{th} %ile	0.015	2.0	0.4	0.3	0.1	2.7
90^{th} %ile	0.025	2.2	0.4	0.7	0.3	3.6
Arithmetic mean	0.021	2.1	0.4	0.5	0.2	3.2

Table 20: Fleet-weighted $PM_{2.5-10}$ brake wear emission factors for the vehicle fleet at Marylebone Road.

	Fleet-weighted $PM_{2.5-10}$ emission factor (mg/km)					
	TWMV	Car	LGV	Rigid HGV/bus	Articulated HGV	All vehicles
2000						
10^{th} %ile	0.046	5.3	1.0	1.3	0.2	7.8
90^{th} %ile	0.071	6.2	1.2	3.5	0.6	11.5
Arithmetic mean	0.061	5.7	1.1	2.8	0.4	10.1
2001						
10^{th} %ile	0.048	5.2	1.0	1.4	0.2	7.8
90^{th} %ile	0.088	6.1	1.2	3.7	0.6	11.7
Arithmetic mean	0.070	5.7	1.1	2.9	0.4	10.2
2002						
10^{th} %ile	0.046	5.3	1.0	1.5	0.2	8.1
90^{th} %ile	0.080	6.1	1.2	3.8	0.6	11.7
Arithmetic mean	0.067	5.7	1.1	2.9	0.4	10.2

Table 21: Fleet-weighted $PM_{2.5-10}$ road surface wear emission factors for the vehicle fleet at Marylebone Road.

	Fleet-weighted $PM_{2.5-10}$ emission factor (mg/km)					
	TWMV	Car	LGV	Rigid HGV/bus	Articulated HGV	All vehicles
2000						
10^{th} %ile	0.019	2.6	0.3	0.8	0.1	3.8
90^{th} %ile	0.027	2.9	0.3	1.9	0.3	5.5
Arithmetic mean	0.023	2.7	0.3	1.5	0.2	4.8
2001						
10^{th} %ile	0.019	2.6	0.3	0.8	0.1	3.9
90^{th} %ile	0.033	2.9	0.3	2.0	0.3	5.6
Arithmetic mean	0.027	2.7	0.3	1.6	0.2	4.9
2002						
10^{th} %ile	0.018	2.6	0.3	0.9	0.1	3.9
90^{th} %ile	0.030	2.9	0.4	2.1	0.3	5.6
Arithmetic mean	0.026	2.7	0.3	1.6	0.2	4.9

Resuspension estimates

Combining abrasion source emissions (Tables 14 and 18) and total PM emissions (Table 10) enables an estimate of resuspension emissions to be made. The emissions due to resuspension alone and resuspension incorporating road surface wear for $PM_{2.5-10}$ and PM_{1-10} are depicted in Tables 22 and 23. Mean values of resuspension emission for $PM_{2.5-10}$ for the years of study range from 1,039 g/km.day to 1,426 g/km.day. For PM_{1-10}, the values range from 791 g/km.day to 1,251 g/km.day.

Table 22: Basic statistics for estimated resuspension emissions for $PM_{2.5-10}$ size fraction at Marylebone Road.

	Resuspension (g/km.day)			Resuspension + road wear (g/km.day)		
	2000	2001	2002	2000	2001	2002
10^{th} %ile	798.3	606.4	500.8	1,139.5	911.6	801.9
90^{th} %ile	1,895.0	1,604.1	1,416.4	2,359.9	2,037.4	1,819.5
Arithmetic mean	1,461.7	1,156.9	1,038.6	1,873.2	1,535.5	1,400.4

Table 23: Basic statistics for estimated resuspension emissions for PM_{1-10} size fraction at Marylebone Road.

	Resuspension (g/km.day)			Resuspension + road wear (g/km.day)		
	2000	2001	2002	2000	2001	2002
10^{th} %ile	492.2	305.2	188.1	831.7	603.0	485.5
90^{th} %ile	1,725.8	1,373.0	1,212.8	2,193.1	1,833.5	1,623.9
Arithmetic mean	1,251.0	917.2	791.0	1,662.5	1,295.9	1,152.8

Comparison of the estimated resuspended component with total non-exhaust emissions suggests that resuspension accounts for 43-49% of the non-exhaust emissions. Furthermore, applying the assumptions that all $PM_{2.5-10}$ emissions originate from non-exhaust sources and that all $PM_{2.5}$ emissions correspond to exhaust emissions, it can be estimated that resuspension emissions are around 30% of the magnitude of exhaust emissions.

The apparent decrease in emissions of resuspended particles in the $PM_{2.5-10}$ size range from 2000 to 2002 reflects the decrease in NO_x emissions predicted from the inventory calculations (Table 8), and highlights the limitation of using a constant $PM_{2.5-10}$:PM_{10} ratio. On the assumption of $PM_{2.5}$ consisting of exhaust emissions and $PM_{2.5-10}$ of non-exhaust emissions, the ratio of $PM_{2.5}$:$PM_{2.5-10}$ should decrease (*i.e.* the ratio of $PM_{2.5-10}$:PM_{10} should increase). Similar estimates for total $PM_{2.5-10}$ and resuspension emissions might then be expected across the three years studied.

Estimates of particle resuspension in the PM_{1-10} portion follow the same trend. However, the magnitudes of the estimates are around 250 g/km.day, smaller than the respective $PM_{2.5-10}$ resuspension emissions. This arises because the increase in abrasion source emissions resulting from the extension of the size fraction from $PM_{2.5-10}$ to PM_{1-10} outweighs the estimated increase in total emissions. The implication is that the PM_{1-10}:PM_{10} ratio of 0.5 used to calculate total PM_{1-10} emissions is too small, and revision of this value is required. Uncertainties in the abrasion source emission factors may also explain this observation.

Apportionment of the resuspended component between light-duty and heavy-duty vehicles

Daily resuspension data and corresponding vehicle count data were sorted by day of the week, and mean values for resuspension for LDVs and HDVs were calculated for weekdays and weekends. The LDVs were taken to comprise cars and LGVs, and excluded motorcycles. The HDV portion consisted of rigid and articulated HGVs, along with buses and coaches. Simultaneous equations were constructed, of the form:

$$\boldsymbol{E_{RESUSP} = xLDV + yHDV} \qquad \text{(Equation 3)}$$

where:

$\boldsymbol{E_{RESUS}}$	is the mean emission due to resuspension
LDV **and** ***HDV***	are the mean daily traffic counts of LDVs and HDVs respectively
$\boldsymbol{x}$ and $\boldsymbol{y}$	are values to be determined by solving the simultaneous equations

The results, along with associated uncertainties are shown in Tables 24 and 25.

Table 24: HDV and LDV emission factors and associated uncertainties for resuspension ($PM_{2.5\text{-}10}$), calculated from simultaneous equations.

Year	Emission factor for resuspension only (mg/km)			
	HDV	±	LDV	±
2000	145	7	2.6	0.8
2001	139	2	0.3	0.4
2002	139	6	-0.5	0.6

Table 25: Combined resuspension and road surface wear emission factors ($PM_{2.5\text{-}10}$) calculated from simultaneous equations.

Year	Emission factor for resuspension and road surface wear (mg/km)			
	HDV	±	LDV	±
2000	162	8	6.0	0.9
2001	157	0	3.8	0.5
2002	156	7	2.9	0.9

Treating resuspension alone (Table 24), the results suggest that HDVs are almost entirely responsible for the resuspension of particles. For HDVs, emission factors for $PM_{2.5\text{-}10}$ due to resuspension range from 139 mg/km to 145 mg/km. This range of values is much smaller than that reported by Abu-Allaban *et al.* (2003) of 230-7800 mg/km, although it should be recognised that the latter findings relate to dusty sites in dry climatic regions, and therefore comparisons with equivalent studies in the UK are likely to be unrepresentative. A mean emission factor for all vehicles of 205 mg/km is presented by Omstedt *et al.* (2005) for a street canyon site in Stockholm. The application of traction sand and studded vehicle tyres in winter months in Sweden results in the emission of large quantities of mechanically generated particles. Consequently, it is not unexpected that the reported emission factor for resuspension in Stockholm is much greater than values derived from the Marylebone Road study.

For the year 2000, the calculated $PM_{2.5\text{-}10}$ emission factor for HDVs (145 mg/km) is more than 150 times larger than the LDV emission factor (2.6 mg/km). Despite the negative value for LDVs calculated for 2002, it is worth noting that within the uncertainty of the calculation, the LDV emission factor could be a small positive value, in which case the ratio of HDV to LDV emission factor would probably be around 200:1.

For the case of combined resuspension and road surface wear emissions (Table 25), LDVs have a greater influence than for resuspension alone. The emission factors derived for HDVs are typically 35 times larger than the LDV emission factors.

Effect of wind speed and direction upon resuspension

It is known that airborne particle concentrations are affected by meteorological variables such as wind direction and speed. Charron and Harrison (2005) reported an observed increase in PM_{coarse} concentration with wind speed based on analysis of data from Marylebone Road. In a study of particulate matter data collected at Hodge Hill, Birmingham, Harrison *et al*. (2004) reported an increase in PM_{coarse}:$PM_{2.5}$ ratio with wind speed. It might, therefore, be expected that resuspension show an increasing trend with wind speed. Wind speed and directional data were available from London Weather Centre and Heathrow Airport, along with more limited data measured at Marylebone Road itself. However, analysis of daily resuspension data revealed no clear wind speed dependency.

It is plausible that, in averaging resuspension to daily values, short-timescale variations of resuspension arising from wind speed effects may be masked by the effects of other influencing factors, particularly traffic-induced effects. To investigate this suggestion, hourly roadside increments of particulate matter and NO_x data were analysed. Total $PM_{2.5-10}$ emissions were calculated based on observed $PM_{2.5-10}$:PM_{10} ratios, as opposed to using a fixed ratio, and values of hourly resuspension were calculated using the same approach as for daily values. PM_{1-10} was excluded from this analysis.

The results of the wind direction and speed analysis are presented in Figures 15 to 17. Figure 15 shows a scatter plot depicting hourly resuspension versus wind direction. To smooth the variability of resuspension due to changes in other controlling factors (*e.g*. traffic intensities) the data were sorted into 1 knot size bins in the case of wind speed and 30° size bins for wind direction, and hourly mean and median values of resuspension were calculated for each size bin. From Figure 16 it can be seen that resuspension appears to show broad peaks in concentration associated with winds in the directions of 60°-90° and 150°-180°, with a third peak centred around the wind direction of 300°.

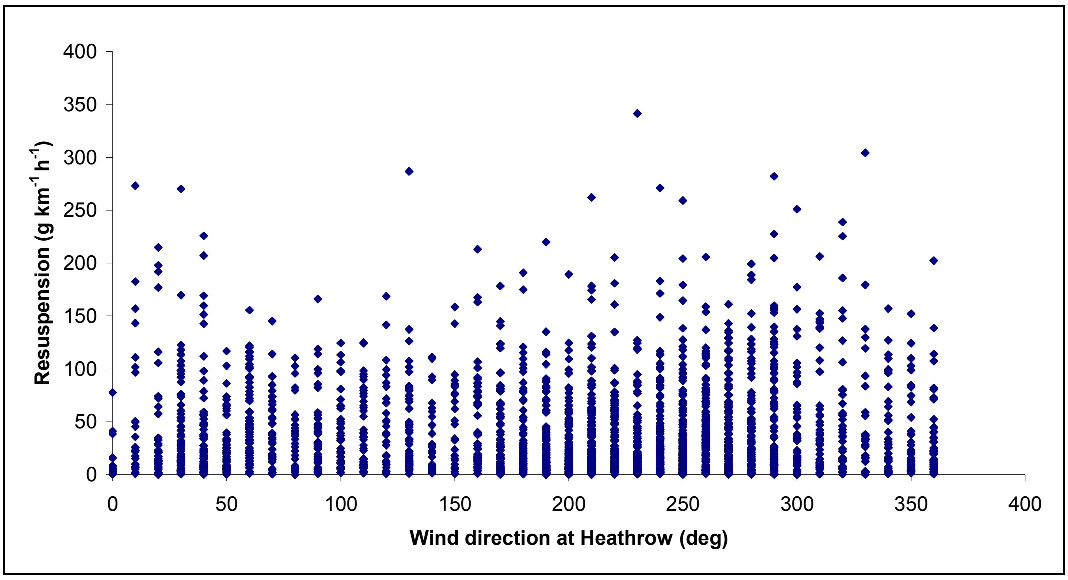

Figure 15: Scatter plot of hourly resuspension at Marylebone Road versus wind direction measured at Heathrow (year=2001).

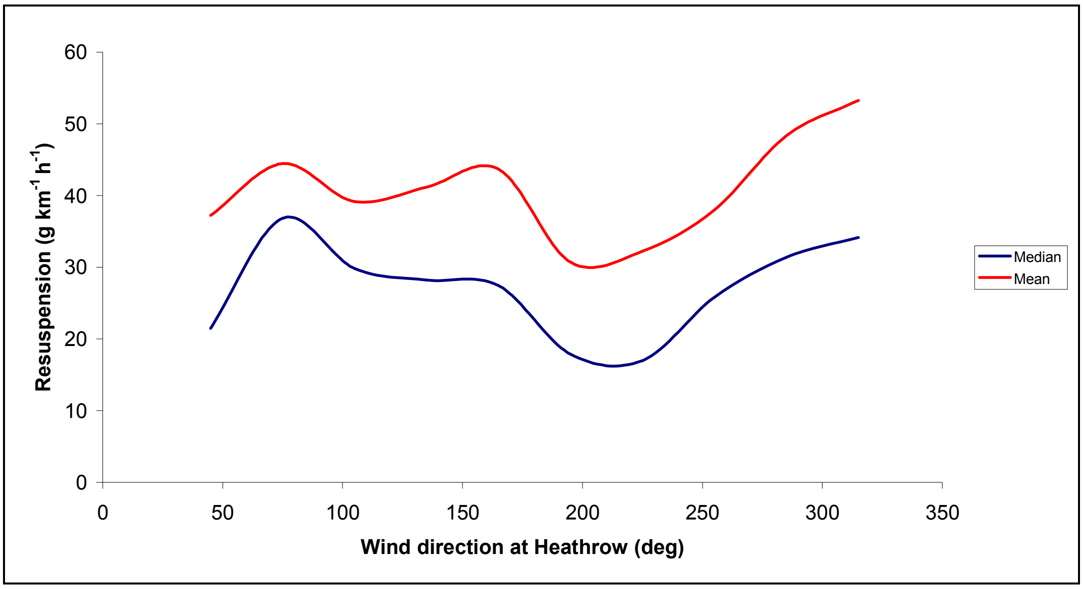

Figure 16: The variation of hourly mean and median resuspension at Marylebone Road with wind direction at Heathrow (year=2001).

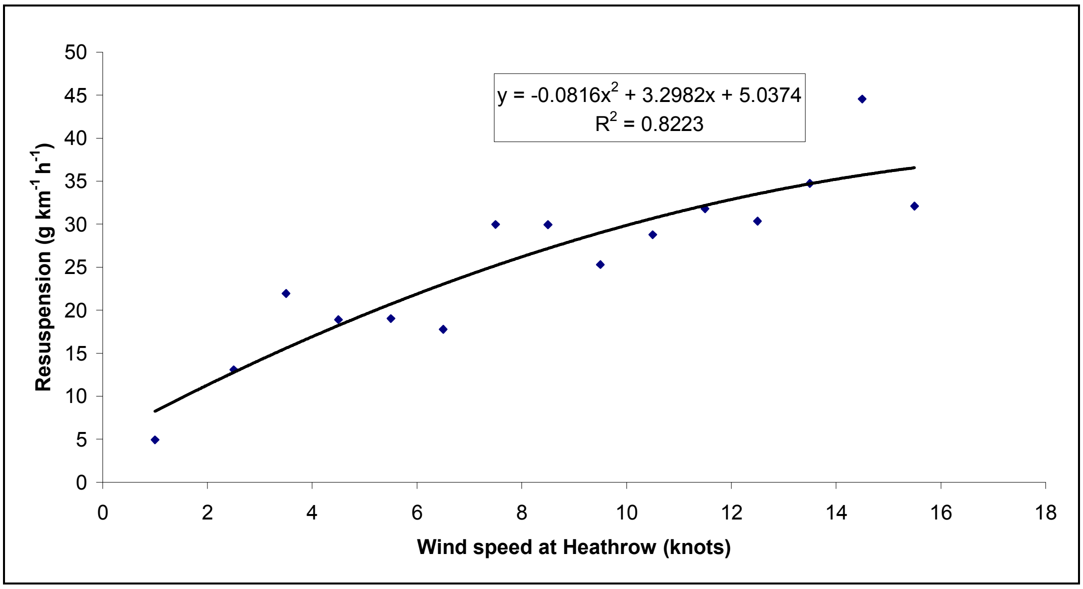

Figure 17: The relationship between hourly median resuspension at Marylebone Road and wind speed at Heathrow (year=2001).

The effect of wind speed on resuspension is highlighted in Figure 17. The data provides evidence of an increase in resuspension with wind speed, with the rate of increase apparently slowing as wind speeds get larger. The fact that the calculated values of resuspension show such a trend with wind speed gives some encouragement that the calculated values are, at least in relative terms, reasonable. However, the apparently large wind speed dependence of resuspension leaves the question of how there is apparently such a strong dependence on heavy-duty vehicles but not light-duty vehicles. We postulate that the key role of the vehicle is in resuspending the particulate matter from the road surface initially, be it by tyre shear or vehicle-generated turbulence. The role of wind speed is in generating the more extensive atmospheric turbulence responsible for keeping the larger particles resuspended such that they then have a significant influence upon airborne concentrations.

Effect of precipitation upon resuspension

Recent work by Omstedt *et al.* (2005) attempted to model the suspension of road dust in Stockholm, Sweden, in terms of the moisture content of the road dust layer, and thus in terms of precipitation. Findings presented by Charron and Harrison (2005) suggest that rain affects both $PM_{2.5}$ and $PM_{2.5\text{-}10}$ concentrations with resulting decreases in measured concentrations after heavy rainfall and higher concentrations during dry periods. The resuspended component of particulate matter might therefore be expected to show a similar trend, with resuspension increasing during dry periods as the road dust layer dries out and becomes more readily suspended. Decreases associated with periods of rainfall arising from runoff, reducing the road dust layer, and hence particles available for resuspension, seems intuitive. Analysis of the Marylebone Road data revealed no clear relationship between resuspension and precipitation. Hourly resuspension data were plotted against hourly precipitation totals in an attempt to reveal any correlations. The precipitation data were offset by one hour to see if rainfall for the previous hour had any effect on resuspension. Neither of these plots revealed any connection between resuspension and precipitation. It appears that simply looking at rainfall totals against resuspension is inadequate for the task of deriving a relationship between the two. Any study of the effects of precipitation should incorporate descriptions of the duration of rainfall events, the rainfall intensity (and hence run-off, which is likely to reduce the dust available for resuspension), and evaporation, as in the model presented by Omstedt *et al.* (2005).

4.2.7 Revised estimates of resuspension using a measured $PM_{2.5}$:PM_{10} ratio

It was noted earlier for Marylebone Road that the apparent decrease in emissions of resuspended particles in the $PM_{2.5\text{-}10}$ size range between 2000 and 2002 may have arisen from the use of a constant $PM_{2.5\text{-}10}$:PM_{10} emissions ratio of 0.4, and reflected the decrease in PM_{10} emissions. A value of 0.4 was chosen based on values cited in the literature for observed atmospheric $PM_{2.5\text{-}10}$:PM_{10} ratios. If the observed temporal decrease (2000-2002) calculated for PM_{10} emissions were the result of a reduction in exhaust emissions of PM, then a change in the $PM_{2.5}$:PM_{10} ratio would be anticipated, assuming no change in non-exhaust emissions. The appropriateness of applying a constant ratio was therefore investigated using a variable $PM_{2.5\text{-}10}$:PM_{10} ratio to calculate total coarse particle emissions and to derive estimates of resuspension.

NO_x emissions were calculated as described earlier. Fleet-weighted emission factors for each vehicle category, along with total daily emission data, were shown in Table 8. Linear regression of PM_{10} against NO_x roadside increment concentrations was used to deduce the relationship between PM_{10} and NO_x emissions. Total PM_{10} and $PM_{2.5\text{-}10}$ emissions for 2000-2002 were presented in Table 10. $PM_{2.5\text{-}10}$ emissions were estimated using annual mean values of observed $PM_{2.5\text{-}10}$:PM_{10} ratios. It was anticipated that the $PM_{2.5\text{-}10}$:PM_{10} ratio would increase between 2000 and 2002, reflecting a reduction in exhaust-related PM emissions in conjunction with the observed reduction in NO_x. The non-exhaust derived fraction would be expected to be unaffected, thus leading to an increase in the $PM_{2.5\text{-}10}$:PM_{10} ratio. The ratios used for the three years are presented in Table 26.

Table 26: Annual mean $PM_{2.5\text{-}10}$:PM_{10} ratios used to estimate coarse particle emissions at Marylebone Road.

Year	$PM_{2.5\text{-}10}$:PM_{10} ratio
2000	0.35
2001	0.32
2002	0.34

Resuspension estimates were calculated by the difference between the abrasion source emissions calculated using the EMEP method (taken from Tables 11-14) and the total coarse particle emission. The resuspension data are presented in Table 27. It is clear from the data presented in Table 27 that the use of observed $PM_{2.5-10}$:PM_{10} ratios to calculate $PM_{2.5-10}$ emissions, and subsequently to estimate resuspension, offers little improvement over the use of the constant ratio. The observed ratios (Table 26), when averaged over an annual timescale show little variation from year to year. However, in contrast to the proposed hypothesis that the observed $PM_{2.5-10}$:PM_{10} ratio would increase year by year as a consequence of reduced exhaust PM emissions, the measured ratios indicate a small decrease from 2000 to 2001, followed by an increase in 2002 to a similar value as 2000. As a result, calculated resuspension emissions follow similar trends to those calculated using a constant ratio. Resuspension emissions were determined to be around 40% lower for 2001 and 2002 when compared with 2000. Since the ratios for each year are smaller than the constant ratio of 0.4 used in the initial calculations (the results of which were given in Table 22), the corresponding resuspension emission estimates are smaller, ranging from 643 g/km.day to 1086 g./km.day.

Table 27: Estimated resuspension emissions for $PM_{2.5-10}$ size fraction at Marylebone Road using variable $PM_{2.5-10}$:PM_{10} ratio.

	Resuspension (g/km.day)			Resuspension + road wear (g/km.day)		
	2000	2001	2002	2000	2001	2002
10^{th} %ile	538.8	251.5	261.7	876.8	559.5	554.9
90^{th} %ile	1,439.1	937.6	976.1	1905.0	1391.1	1382.3
Arithmetic mean	1,086.8	642.6	679.7	1498.3	1021.3	1041.5

Due to the unexpected decreasing trend in the $PM_{2.5-10}$: PM_{10} ratio, a further investigation was undertaken of the trends in the measured concentrations of the three PM metrics at the Marylebone Road and Bloomsbury sites, and the trend in the roadside increment.

4.2.8 Analysis of temporal PM_{10}, $PM_{2.5}$, and $PM_{2.5-10}$ trends

Since the $PM_{2.5-10}$:PM_{10} ratio of the roadside increment PM was observed not to follow the anticipated increasing trend from 2000 to 2002, an investigation was carried out to examine the temporal trends of the various particle metrics. Data were compiled for all available years (1998-2004), with monthly mean PM_{10}, $PM_{2.5}$, and $PM_{2.5-10}$ concentrations being calculated for Marylebone Road, Bloomsbury, and roadside increment. The $PM_{2.5-10}$:PM_{10} ratios were calculated using the monthly mean data.

PM_{10}, $PM_{2.5}$, and $PM_{2.5-10}$ concentrations

Plots of the monthly mean PM_{10}, $PM_{2.5}$, and $PM_{2.5-10}$ concentrations between 1998 and 2004 for Marylebone Road, Bloomsbury, and the roadside increment are shown in Figures 18 to 20. Six-month moving average trend lines are fitted to the data to reveal longer-term variations in concentrations. The trends for Marylebone Road are shown in Figure 18. Examining the trend in PM_{10} reveals a range of between 40 $\mu g/m^3$ and around 52 $\mu g/m^3$, rising from a minimum around March 1999 to a peak between October 1999 and February 2000. Concentrations then fell gradually to September 2001, before a small, sharp rise in November and December 2001. Following a small decrease, PM_{10} concentrations then appear to have risen steadily to a maximum of around 50 $\mu g/m^3$ in July 2003.

Analysing the time series of the other particle metrics reveals possible sources contributing to episodes of elevated PM_{10} concentration. A coincident peak in $PM_{2.5-10}$ concentrations between October 1999 and April 2000 suggests that local construction activity, resulting in the generation of a higher proportion of $PM_{2.5-10}$, may have been responsible for elevated PM_{10}. Alternatively, since this peak occurs during the autumn and winter period, elevations due to use of road salt, enhancing the road dust layer available for resuspension, could be a plausible mechanism. The smaller peak in PM_{10} between December 2001 and March 2002 correlates well with an increase in $PM_{2.5}$ concentrations, indicating a combustion-related process or contribution of secondary particles.

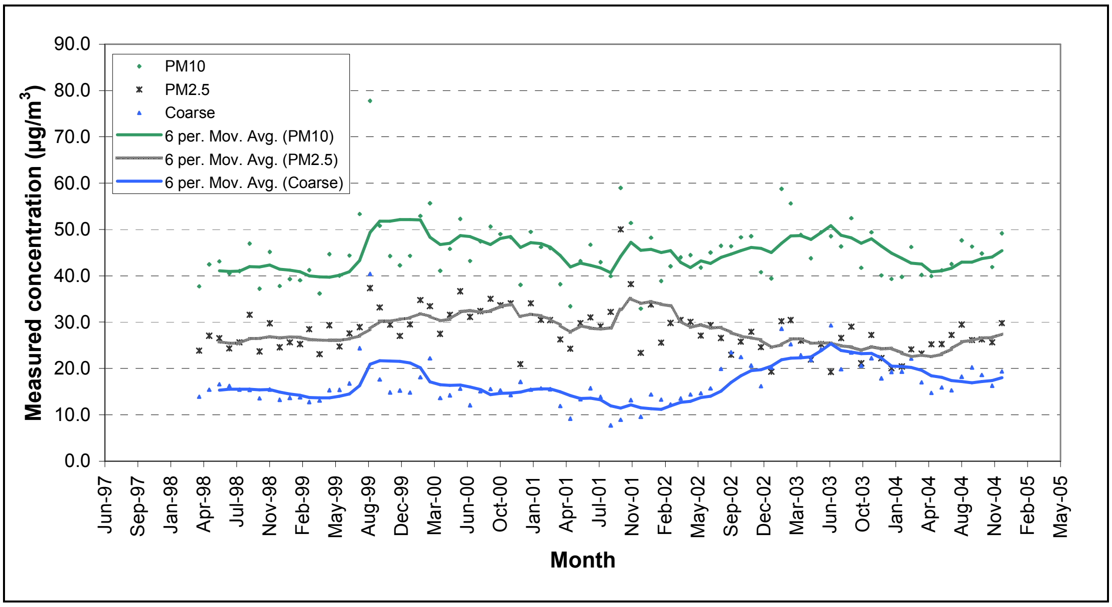

Figure 18: Trends (6-month moving average) in the concentrations of the three PM metrics at Marylebone Road between January 1998 and December 2004.

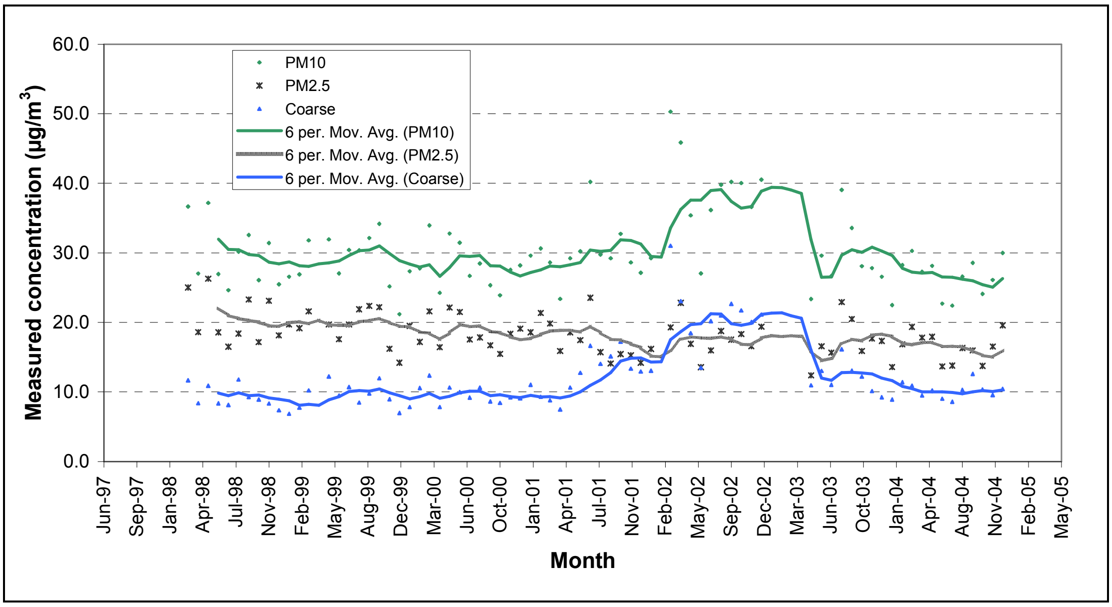

Figure 19: Trends (6-month moving average) in the concentrations of the three PM metrics at Bloomsbury between January 1998 and December 2004.

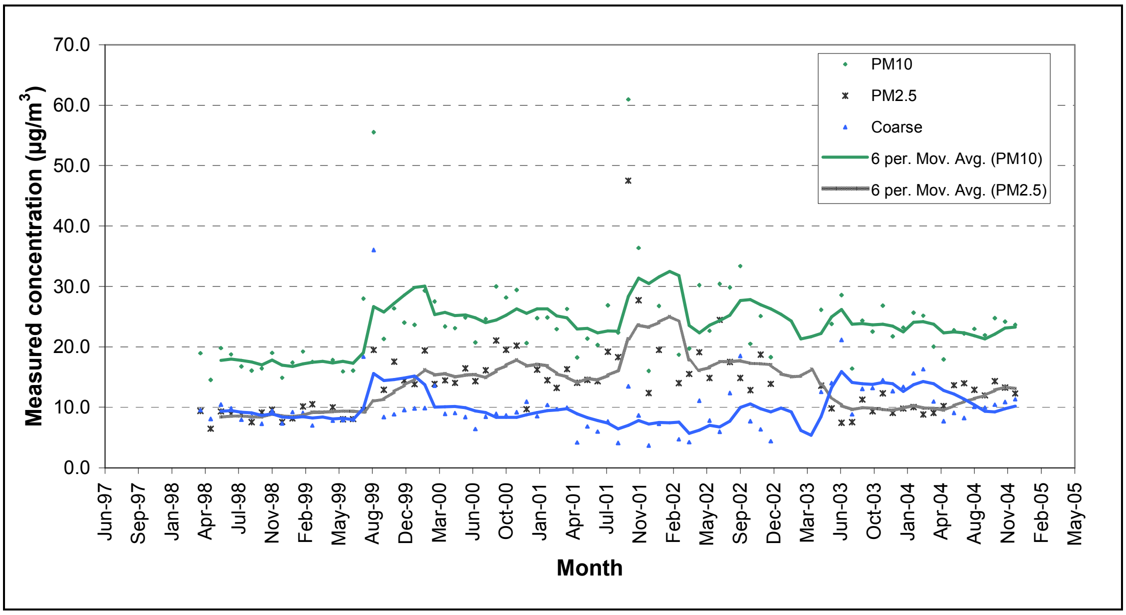

Figure 20: Trends in the concentrations of the three PM metrics in the roadside increment (Marylebone Road – London Bloomsbury) between January 1998 and December 2004.

From March 2002 $PM_{2.5}$ concentrations at Marylebone Road are seen to decrease to values similar to those in 1998 and the first half of 1999 (around 25 µg/m^3). Despite this, PM_{10} concentrations increase due to a doubling of the coarse particle concentration, from around 12 µg/m^3 to 25 µg/m^3.

Examination of PM data from the Bloomsbury urban background site (Figure 19) and the roadside increment (Figure 20) was performed to try and elucidate the causes for the trends observed in the Marylebone Road roadside data. Comparing Figure 18 and Figure 19 it is apparent that the observed peak in PM_{10} at Marylebone Road between October 1999 and February 2000 is absent in the Bloomsbury time series, which suggests the cause of the increase at Marylebone Road was a local rather than regional phenomenon. This provides evidence for either traffic-related non-exhaust sources or construction work in the vicinity of Marylebone Road. From January 1998 to April 2001, PM_{10}, $PM_{2.5}$, and $PM_{2.5\text{-}10}$ all display approximately constant levels. After April 2001 $PM_{2.5}$ concentrations are observed to fall by 2-3 µg/m^3. However, PM_{10} concentrations rise by around 8-10 µg/m^3 to a broad peak from February 2002 to March 2003. The fact that this trend is mirrored in the coarse particle concentration is evidence that the Bloomsbury site was subject to a source of coarse PM during this time period. The gradual increase of PM_{10} and $PM_{2.5\text{-}10}$ at Marylebone Road over the same period indicates a source of regional importance and so is unlikely to arise from traffic-generated emissions.

Roadside incremental concentrations (Figure 20) display somewhat more variability than either Marylebone Road or Bloomsbury. There are, however, a number of notable points. Firstly, the peak seen in the Marylebone Road PM_{10} and $PM_{2.5\text{-}10}$ data during the winter months of 1999 to 2000 is manifest in the roadside increment data. Six-month moving average PM_{10} concentrations increase from around 18 µg/m^3 in July 1999, peaking at 30 µg/m^3 in January and February 2000. This provides further evidence of either non-exhaust traffic emissions or construction activity on or near to Marylebone Road.

Comparison of Figure 18 and Figure 20 indicates the September 2001 to March 2002 maximum in the Marylebone Road data is also apparent in the roadside increment data. It is important to note that during this same time period $PM_{2.5\text{-}10}$ concentrations in the roadside increment are constant whilst the $PM_{2.5}$ fraction follows the trend in PM_{10}. This observation is indicative of emissions from combustion processes or secondary particles. The absence of a concentration increase during the same months at Bloomsbury suggests a traffic-related emission source rather than emissions arising from domestic or industrial combustion as such emissions may be expected to affect a greater area and hence be apparent in the urban background data.

The steady increase in PM_{10} and $PM_{2.5\text{-}10}$ seen at Marylebone Road between April 2002 and April 2003 is not repeated in the roadside increment trends. An interpretation of this may be that concentrations at Bloomsbury were particularly high during much of this period, thus offsetting any increase in the roadside increment. Secondly, missing data between January and April 2003 in the Bloomsbury dataset, and therefore in the roadside increment data, may have the effect of extending the PM_{10} and $PM_{2.5\text{-}10}$ peaks in the Bloomsbury data. This would result in unrepresentative trends in the roadside increment over this time period.

A final feature of the roadside increment data is the switch in predominance of the $PM_{2.5}$ and $PM_{2.5\text{-}10}$ size fractions after May 2003. Prior to this date, PM_{10} was composed predominantly of $PM_{2.5}$, apart from the latter six months of 1999. However, after May 2003, $PM_{2.5\text{-}10}$ became the dominant fraction of PM_{10} in the roadside increment. This observation, and other trends in the proportion of PM_{10} composed of $PM_{2.5}$ and $PM_{2.5\text{-}10}$, are discussed below.

$PM_{2.5\text{-}10}$:PM_{10} ratios

Monthly mean $PM_{2.5\text{-}10}$:PM_{10} ratios for Marylebone Road, Bloomsbury, and the roadside increment are depicted in Figures 21 to 23. Six-month moving average trend lines have again been fitted to the data to clarify any inherent trends.

Inspection of the trend for Marylebone Road (Figure 21) reveals a decrease in the $PM_{2.5\text{-}10}$:PM_{10} ratio from October 1999 to January 2002, from a maximum of 0.40 to a minimum of around 0.27. Particularly low values were observed for September and October 2001 (0.19 and 0.15, respectively). This correlates with the increase in PM_{10} and $PM_{2.5}$ concentrations in the Marylebone Road and roadside increment data (Figures 18 and 20). During 2002 and the first half of 2003, the ratio increases progressively to around 0.50 in July 2003. After this time the ratio falls again before assuming a constant value of around 0.40 in the latter part of 2004. In contrast to the anticipated decrease in the $PM_{2.5\text{-}10}$:PM_{10} ratio with time, due to the reduction in exhaust PM emissions, there appears to be no constant trend in the observed ratio for Marylebone Road. Over the time period used in the estimation of resuspension emissions the ratio shows a tendency to decrease.

$PM_{2.5\text{-}10}$:PM_{10} ratios for Bloomsbury are displayed in Figure 22. The ratio remains rather static from 1998 to the end of 2000, with a typical value of 0.35. In the latter half of 2001 and throughout 2002 the ratio increases markedly to around 0.50, indicative of construction activity close to the monitoring site resulting in the emission of coarse PM. The ratio is reduced gradually during 2003 to a steady level of 0.38 – 0.40.

The decreasing trend of the proportion of PM_{10} composed of coarse PM in the Marylebone Road data is accentuated in the roadside increment plot (Figure 23). From a maximum value of greater than 0.50 around September 1999, the ratio is observed to fall substantially to less than 0.25 by the end of 2001. This is a reflection of the large increase in $PM_{2.5\text{-}10}$:PM_{10} ratio at Bloomsbury throughout 2001 particularly, and highlights that during these months the use of Bloomsbury as a background site by which to calculate roadside increments may be questionable. The trend reverses in 2002, and taking into consideration the four months of missing data at the beginning of 2003, the proportion of coarse PM appears to rise continually to a value in excess of 0.55. In 2004, the ratio decreases, settling at a value of around 0.42.

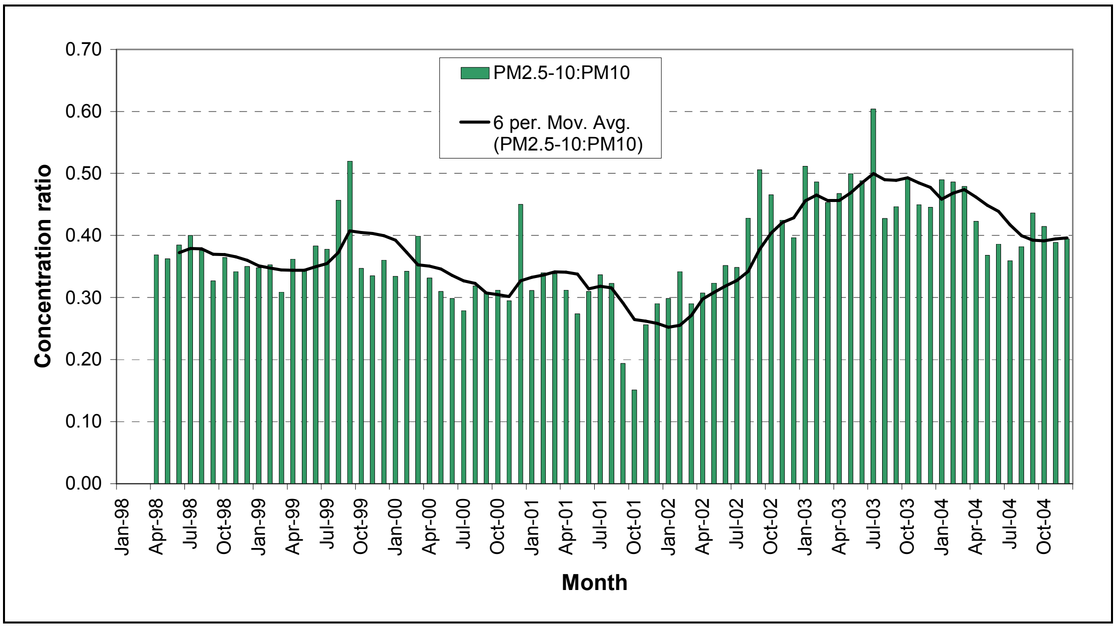

Figure 21: Trend in the $PM_{2.5\text{-}10}$:PM_{10} ratio at Marylebone Road between January 1998 and December 2004 (monthly mean and 6-month moving average).

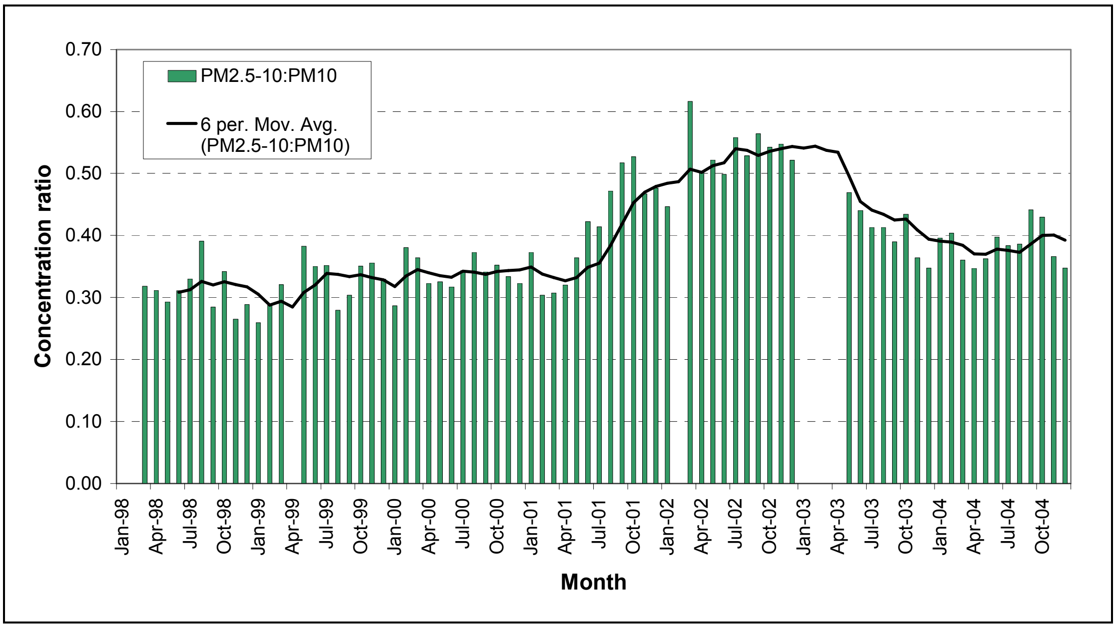

Figure 22: Trend in the $PM_{2.5\text{-}10}$:PM_{10} ratio at London Bloomsbury between January 1998 and December 2004 (monthly mean and 6-month moving average).

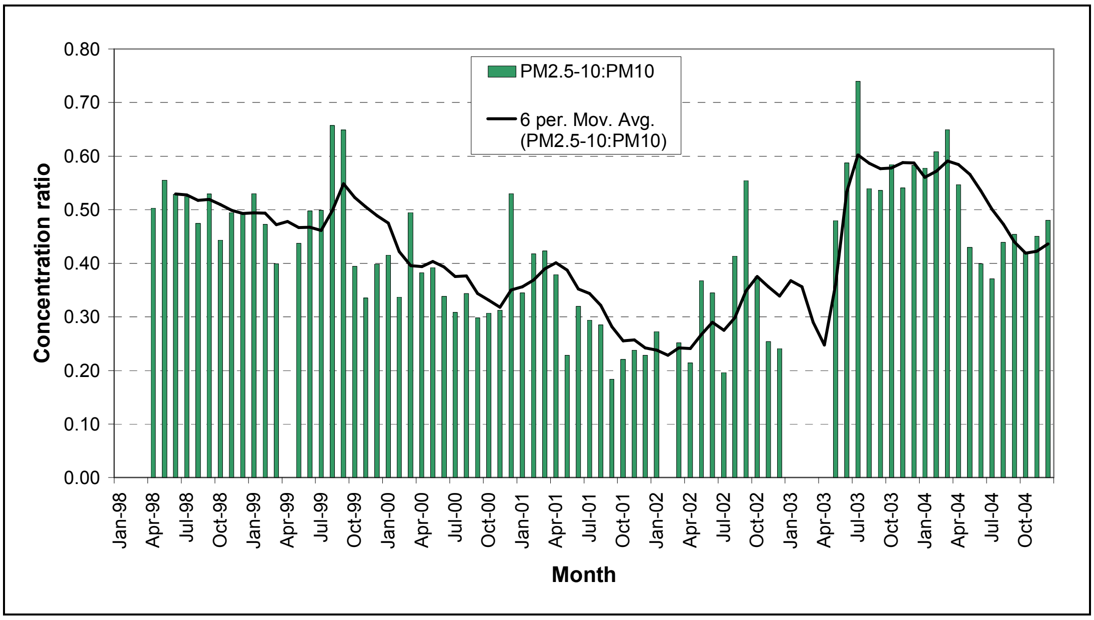

Figure 23: Trend in the $PM_{2.5-10}$:PM_{10} ratio in the roadside increment (Marylebone Road – London Bloomsbury) between January 1998 and December 2004 (monthly mean and 6-month moving average).

It can be concluded from the observations discussed above that the anticipated increase in the ratio $PM_{2.5-10}$:PM_{10} is not conclusively supported by data from Marylebone Road or the roadside increment data. It is also apparent that the selection of appropriate background sites for calculation of roadside incremental concentrations is imperative. The effect of local sources upon concentrations measured at London Bloomsbury may go some way to explaining the unexpected trends in the proportion PM_{10} composed of coarse PM in the roadside increment at Marylebone Road.

4.3 Comparison of EMEP and HDM-4 methods for tyre wear PM_{10} emissions

Using traffic data collected at Marylebone Road between 1 January 2000 and 31 December 2002, a comparison of two different methods for estimating the contribution to particulate matter emissions was carried out. It was seen in earlier work that estimates of PM emissions due to abrasion sources vary depending on the modelling approach used. Further to this, it was decided to compare the figures obtained using the EMEP/CORINAIR method, which was used in the calculation of the contribution of resuspension at Marylebone Road, and the HDM-4 tyre wear model.

The EMEP/CORINAIR method is based on a relatively small number of studies that produced a range of estimates for emission factors (Boulter, 2005b, and references therein). The method incorporates emission factors for eight vehicle categories:

- Two-wheeled motor vehicles (TWMV)
- Cars
- Light goods vehicles (LGV)
- Two-axle heavy-goods vehicles (HGV) and coaches
- Three-axle HGVs and coaches
- Four-axle HGVs
- Five-axle HGVs
- Six-axle HGVs.

In the case of tyre wear, a correction factor for vehicle speeds between 40 km/h and 90 km/h is included. Below 40 km/h and above 90 km/h the values are assumed constant. In addition to the speed correction, the EMEP/CORINAIR approach provides mass fractions for the size metrics PM_{10}, $PM_{2.5}$, PM_1, and $PM_{0.1}$, and, in the case of HGVs, a correction for vehicle load.

The HDM-4 tyre consumption model is a mechanistic approach to the problem of quantifying tyre wear. It considers the key factors that influence tyre wear to be road surface roughness, the severity of the road alignment, in particular the horizontal curvature, acceleration and deceleration of vehicles, vehicle loading, and properties of the tyre, such as whether the tyre is new or a retread, properties of the rubber, and inflation pressure, for example (Bennett and Greenwood, 2003). The model describes tyre wear in terms of speed variation, mass of the vehicle and vehicle loading, and circumferential and lateral forces on the tyre (Carpenter and Cenek, 1999). The model also incorporates factors to take into consideration local effects, including road surface roughness and macrotexture, temperature and humidity factors, vehicle maintenance, and driving style. However, due to a lack of detailed information required, such factors are assigned default values in the current model (Bennett & Greenwood, 2003). In this project, tyre wear emission factors for PM_{10} have been included as an adaptation to the HDM-4 method, based on the rather crude assumption that 5% of total tyre wear actually becomes airborne PM.

4.3.1 Method

The method of estimating tyre wear emissions using the EMEP/CORINAIR procedure can be found in Appendix B. The only modification made was to calculate PM_{10} emissions, as opposed to the $PM_{2.5\text{-}10}$ emissions calculated previously, so a direct comparison of the emissions estimates using the two models could be made. Calculations of tyre wear PM_{10} emissions using HDM-4 were carried out using the speed variation set to medium, and a road radius of 425 m. Despite Marylebone Road being a straight stretch of road, a value of 425 m was used to consider the effect of vehicles changing lanes.

Emission factors were calculated using vehicle speeds between 25 km/h and 60 km/h for each vehicle category. Since the categories given in the HDM-4 model differ from those in the Marylebone Road dataset, two assumptions were made. Firstly, it was assumed that all rigid HGVs, coaches, and buses at Marylebone Road had three axles, and secondly, that all articulated HGVs had five axles. Plots were then constructed of tyre wear emission factor (g/km) versus mean vehicle speed (km/h) for each vehicle category and polynomial trend lines fitted to the data (Figure 24).

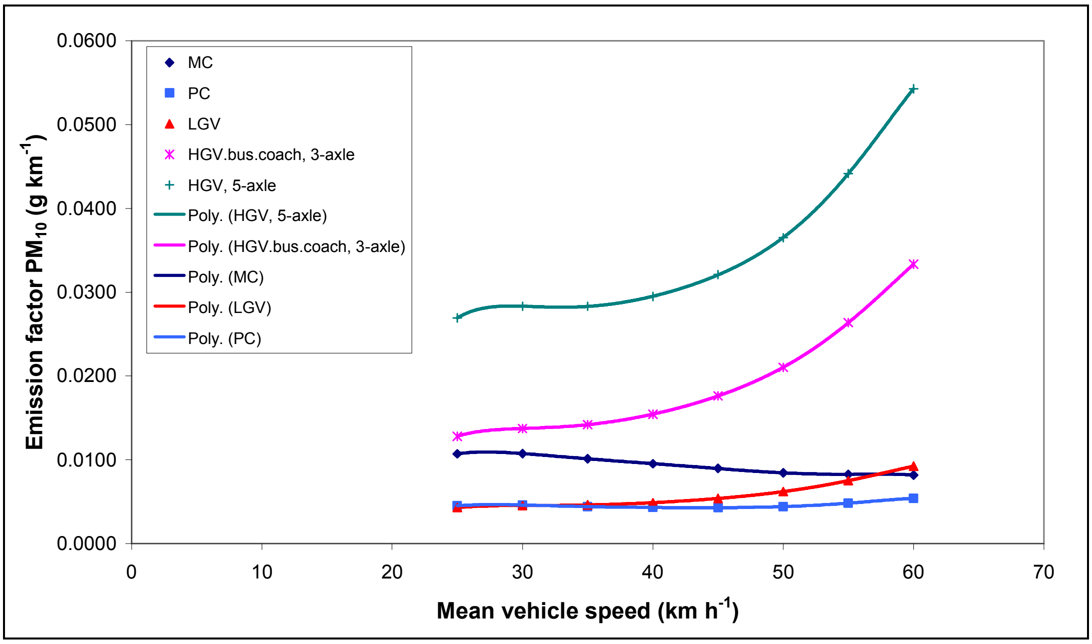

Figure 24: Relationships between mean vehicle speed and tyre wear PM_{10} emission factors for the main vehicle categories (medium speed variation and 435 m road radius).

The equations from the trend lines were combined with measured daily-mean vehicle speed at Marylebone Road to derive emission factors for each vehicle category. The emission factors obtained for each vehicle class were then multiplied by the observed number of vehicles (vehicles per day) passing along Marylebone Road for the corresponding day to obtain the tyre wear emissions (g/km.day) from each category. The sum of the emissions from all vehicle categories provided an estimate of total tyre wear emissions.

4.3.2 Results

Tyre wear PM_{10} emissions calculated using the HDM-4 model and the EMEP/CORINAIR procedure are presented in Tables 28 and 29. The data indicate that the HDM-4 method yields daily tyre wear emissions that are on average around 56% of the values produced by the EMEP/CORINAIR model.

Further analysis of the data reveals that both models predict a similar decrease in tyre wear emissions from 2000 to 2002, with mean-daily emissions decreasing by 11% in the case of the HDM-4 model, and 12% when the EMEP/CORINAIR procedure is applied.

The HDM-4 model suggests that emissions from TWMVs are of greater importance than EMEP/CORINAIR; TWMVs are responsible for 2.9 – 3.3% of tyre wear emissions according to HDM-4 compared to 0.6 – 0.7% in the case of EMEP/CORINAIR. However, in terms of the contributions of light-duty and heavy-duty vehicles overall, EMEP/CORINAIR attributes a greater share of the total emissions (~78%) to light-duty vehicles than HDM-4 (~70%).

Table 28: Summary of annual mean-daily tyre wear PM_{10} emissions at Marylebone Road calculated using the HDM-4 model (values based on medium speed variation, road radius of 425 m).

Year and statistic	Daily PM_{10} emission (g/km.day) due to tyre wear (HDM-4)							
	TWMV	Car	LGV	Rigid HGV/bus	Artic. HGV	Total LDV	Total HDV	Total
2000								
10^{th} %ile	10.1	264.7	36.6	58.4	13.2	315.3	73.3	416.6
90^{th} %ile	17.1	318.2	44.0	145.0	46.6	375.8	189.2	561.9
Arithmetic mean	14.3	295.5	39.9	116.3	32.2	349.7	148.6	498.3
% of total	2.87%	59.3%	8.01%	23.35%	6.47%	70.18%	29.82%	100.00%
2001								
10^{th} %ile	9.8	246.8	34.0	57.3	13.5	296.8	73.4	390.3
90^{th} %ile	21.1	307.5	42.8	146.5	44.2	367.1	188.1	553.7
Arithmetic mean	15.8	278.3	38.1	113.8	30.4	332.3	144.2	476.5
% of total	3.32%	58.41%	8.00%	23.89%	6.38%	69.73%	30.27%	100.00%
2002								
10^{th} %ile	9.1	263.3	32.4	56.4	14.5	282.9	72.8	380.5
90^{th} %ile	16.9	277.9	38.8	134.5	40.8	329.7	172.6	493.7
Arithmetic mean	13.9	259.2	35.5	106.0	29.2	308.5	135.2	443.7
% of total	3.14%	58.41%	7.99%	23.88%	6.59%	69.53%	30.47%	100.00%

Table 29: Summary of annual mean-daily tyre wear PM_{10} emissions at Marylebone Road calculated using the EMEP/CORINAIR approach.

Year and statistic	Daily PM_{10} emission (g/km.day) due to tyre wear (EMEP)							
	TWMV	Car	LGV	Rigid HGV/bus	Artic. HGV	Total LDV	Total HDV	Total
2000								
10th %ile	4.1	523.7	100.2	65.1	19.2	628.0	84.3	712.4
90th %ile	6.5	641.4	122.8	183.9	76.3	770.6	260.1	1,030.8
Arithmetic mean	5.5	592.6	113.4	142.2	51.3	711.5	193.4	905.0
% of total	*0.61%*	*65.48%*	*12.53%*	*15.71%*	*5.66%*	*78.62%*	*21.38%*	*100.00%*
2001								
10th %ile	3.5	428.0	82.6	63.1	19.3	514.1	82.4	596.5
90th %ile	8.1	622.8	120.3	183.7	71.8	751.1	255.5	1,006.6
Arithmetic mean	5.9	543.6	105.0	135.1	47.5	654.5	182.7	837.1
% of total	*0.71%*	*64.94%*	*12.54%*	*16.14%*	*5.68%*	*78.18%*	*21.82%*	*100.00%*
2002								
10th %ile	3.5	469.0	90.9	62.9	21.2	563.4	84.1	647.5
90th %ile	6.5	561.9	108.9	169.3	66.2	677.3	235.5	912.9
Arithmetic mean	5.3	515.5	99.9	129.0	46.1	620.7	175.1	795.8
% of total	*0.66%*	*64.78%*	*12.56%*	*16.21%*	*5.79%*	*78.00%*	*22.00%*	*100.00%*

4.4 Calculation of non-exhaust PM_{10} emissions using German traffic situation model

A version of the USEPA AP-42 model to estimate particulate matter emissions was adapted for use in Berlin (Rauterberg-Wulff, 2000, cited in Boulter, 2005b), with subsequent modifications to separate the exhaust and non-exhaust components by Düring *et al.* (2002). The latter was found to over-estimate emissions (Lohmeyer *et al.,* 2003, cited in Ketzel *et al.,* 2005; Lohmeyer *et al.,* 2004b), and the modified AP-42 model approach was replaced by the present traffic situation method.

The procedure is based on the combined use of emission factors for typical vehicles, taken from the INFRAS Emissions Handbook (INFRAS, 2004, cited in Boulter, 2005b), and the method detailed in Gehrig *et al.* (2003) describing the variation of these emission factors with different traffic regimes. Gehrig *et al.* (2003) distinguished between exhaust and non-exhaust emissions on the basis of PM_{10} and PM_1 measurements and the assumption that PM_1 represents direct exhaust emissions and PM_{1-10} the contribution from abrasion and resuspension. The results obtained were compared with the calculated $PM_{2.5-10}$ (*i.e.* non-exhaust) emissions for Marylebone Road to determine the differences that result from using these two approaches.

4.4.1 Method

Calculations of non-exhaust PM_{10} emissions by vehicle category using hourly traffic count and mean vehicle speed data collected at Marylebone Road were performed using the German traffic situation model. Based on the configuration of Marylebone Road, the emission factors corresponding to the 'main road with traffic lights' traffic situation (Table 30) were used initially to estimate non-exhaust PM_{10} emissions. To account for delays, mean vehicle speed was used as a guide. Hours when mean vehicle speed was less than 35 km/h were taken to represent times of heavy delays, between 35 km/h and 45 km/h medium delays, and over 45 km/h minimal delays.

Table 30: Emission factors (mg/vkm) used in applying the traffic situation model to the Marylebone Road traffic dataset.

Traffic situation	Speed limit (km/h)	% constant speed driving	Total non-exhaust PM_{10} emission factor (mg/vkm)				
			V1	V2	V3	V5	V7
			TWMV	Car	LGV	3-axle HGV, bus, coach	5-axle HGV
Main road, traffic lights, minimum delay	50	44	40	40	40	380	380
Main road, traffic lights, medium delay	50	32	60	60	60	600	600
Main road, traffic lights, heavy delay	50	28	90	90	90	800	800
Main road, right of way, minimal hold ups	50	52	30	30	30	300	300
Main road, right of way, medium hold ups	50	44	40	40	40	380	380
Main road, right of way, major hold ups	50	37	50	50	50	450	450

Emissions for two-wheeled motor vehicles, cars, and LGVs were then added together to obtain hourly figures for emissions due to light-duty vehicles; emissions from buses, rigid HGVs, and articulated HGVs were likewise summed to derive equivalent figures for heavy-duty vehicles. The sum of emissions from all vehicle classes gave total non-exhaust emissions. Daily values were calculated for days with a complete set of twenty-four hourly values. Additional calculations were then carried out using this method, but applying the 'main road, right of way' traffic situation emission factors.

4.4.2 Results

Main road with traffic lights

Summary statistics of daily non-exhaust PM_{10} emissions light- and heavy-duty vehicle categories, along with total emissions obtained using traffic situation model are presented in Table 31. Compared with PM emissions data obtained using NO_x:PM_{10} relationships and NO_x emissions for Marylebone Road (Table 10) it can be seen that the traffic situation model results in non-exhaust PM_{10} emissions that are around 55% larger than the *total* PM_{10} emissions calculated for Marylebone Road. Compared with the calculated non-exhaust PM emissions at Marylebone Road ($PM_{2.5\text{-}10}$ emissions), the traffic situation model produces values almost five times as large.

Table 31: Summary statistics of annual mean-daily non-exhaust PM_{10} emissions at Marylebone Road calculated for the traffic situation 'main road with traffic lights'.

	Daily PM_{10} non-exhaust emissions (g/km.day)								
	LDV			HDV			Total		
	2000	2001	2002	2000	2001	2002	2000	2001	2002
10^{th} %ile	4,174.2	3,161.3	3,921.6	2,297.5	2,252.3	2,445.9	6,720.1	6,153.3	6,707.4
90^{th} %ile	6,139.7	5,893.3	5,336.7	8,086.5	7,834.4	7,296.7	14,207.9	13,686.3	12,566.5
Arith.mean	5,183.5	4,807.2	4,672.9	5,820.6	5,523.5	5,354.6	11,004.2	10,330.7	10,027.5
% of total	*47.11%*	*46.53%*	*46.60%*	*52.89%*	*53.47%*	*53.40%*	*100.00%*	*100.00%*	*100.00%*

Since the traffic situation approach incorporates the use of the $PM_{1\text{-}10}$ size fraction as representing non-exhaust PM and our method uses $PM_{2.5\text{-}10}$ size fraction, some discrepancy between estimates should be expected. However, even correcting our emissions to reflect the additional non-exhaust emissions in the $PM_{1-2.5}$ fraction (Table 10, PM_{1-10} emissions) still produces emissions that are around three times smaller than those predicted by the traffic situation model.

Examination of the inter-annual trends reveals that the traffic situation model suggests a decrease in annual mean-daily emissions between 2000 and 2002, and since the same emission factors are used throughout, this must reflect a change in the traffic flow, vehicle fleet composition, and driving characteristics.

Main road, right of way

The non-exhaust PM_{10} emissions calculated using the 'main road, right of way' traffic situation with the Marylebone Road data are presented in Table 32. Comparing the values in Tables 31 and 32 it can be seen that the use of these emission factors results in daily non-exhaust PM_{10} emissions 38% - 40% lower than emissions calculated using the 'main road with traffic lights' traffic situation. The values obtained are comparable to the total PM_{10} emissions derived from NO_x:PM_{10} relationships and NO_x emissions (Table 10) based on measurements at Marylebone Road. However, as with the previous situation, when comparing the results from the traffic situation model with our calculated non-exhaust emissions the emissions obtained using the traffic situation model are appreciably larger. Values derived from the traffic situation model were, on average, 141% larger than the $PM_{2.5-10}$ emissions calculated using NO_x as a tracer, and around 94% larger than the corresponding PM_{1-10} values.

Table 32: Summary statistics of annual mean-daily non-exhaust PM_{10} emissions at Marylebone Road calculated for the traffic situation 'main road, right of way'.

	Daily PM_{10} non-exhaust emissions (g/km.day)								
	LDV			HDV			Total		
	2000	2001	2002	2000	2001	2002	2000	2001	2002
10^{th} %ile	2753.9	2222.7	2589.4	1478.4	1499.6	1590.1	4428.4	3852.0	4329.7
90^{th} %ile	3616.3	3505.7	3162.9	4758.5	4625.2	4304.3	8341.8	8122.8	7385.5
Arith.mean	3216.9	2975.9	2874.3	3519.6	3338.1	3228.1	6736.5	6314.0	6102.4
% of total	47.75%	47.13%	47.10%	52.25%	52.87%	52.90%	*100.00%*	*100.00%*	*100.00%*

These findings imply that for the case of estimating non-exhaust emissions at Marylebone Road the traffic situation model produces values that are too large, indeed in excess of the particle source strength itself. This is likely to be a consequence of applying emission factors calculated from research on German roads to a road in a different environment. Furthermore, it should be recognised that the threshold values of vehicle speed used to distinguish between different levels of congestion and hold-ups were assigned rather arbitrarily, and modifying these boundaries may result in emissions closer in magnitude to those derived from NO_x:PM_{10} relationships and NO_x emissions.

4.5 Analysis of TRAMAQ data

During the Project Number UG250 funded under the TRAMAQ Programme, ERM and the University of Birmingham collected 24-hour average data for fine and coarse particle fractions at four sets of sites, each of which comprised a roadside site and a nearby urban background site. The specific locations are described by Harrison *et al.* (2004) and were as follows:

- Elephant and Castle, London
- High Holborn, London

- Park Lane, London
- Selly Oak, Birmingham

The sites differed very significantly in their characteristics with respect not only to traffic mix but also to the degree of openness of the site. High Holborn and Elephant and Castle were both in canyons enclosed by high buildings, whilst Park Lane and Selly Oak had buildings to one side but not the other. In the case of Park Lane, the buildings to one side were very tall and this led to completely different dispersion characteristics for easterly and westerly winds (the road runs approximately north-south). Data for this road are therefore presented disaggregated according to wind direction within two sectors, 0-140º and 140-360º.

The TRAMAQ data could not be subjected to analysis in the same manner as the Marylebone Road data for two reasons:

- The only measurements were of particulate matter mass and chemical composition with no accompanying measurements of NO_x.
- The available traffic data were not measured continuously as at Marylebone Road, but were based upon periodic surveys.

It has therefore been necessary to carry out a more constrained analysis of the data, primarily to investigate its consistency with the conclusions drawn from Marylebone Road.

From the results obtained at Marylebone Road it would be anticipated that the roadside increment of $PM_{2.5\text{-}10}$ would scale according to the volume of heavy-duty vehicles. There was a general upward trend for coarse particle incremental concentration with total traffic, illustrated in Figure 25. However, this figure takes no account of the differing dispersion conditions of the various sites in terms of either their relative openness or the wind speed conditions experienced during sampling which was not conducted simultaneously at all sites. Despite this, $PM_{2.5}$ does show a relatively linear increase with diesel traffic volume when the data for the two wind sectors at Park Lane are averaged (see Figure 26).

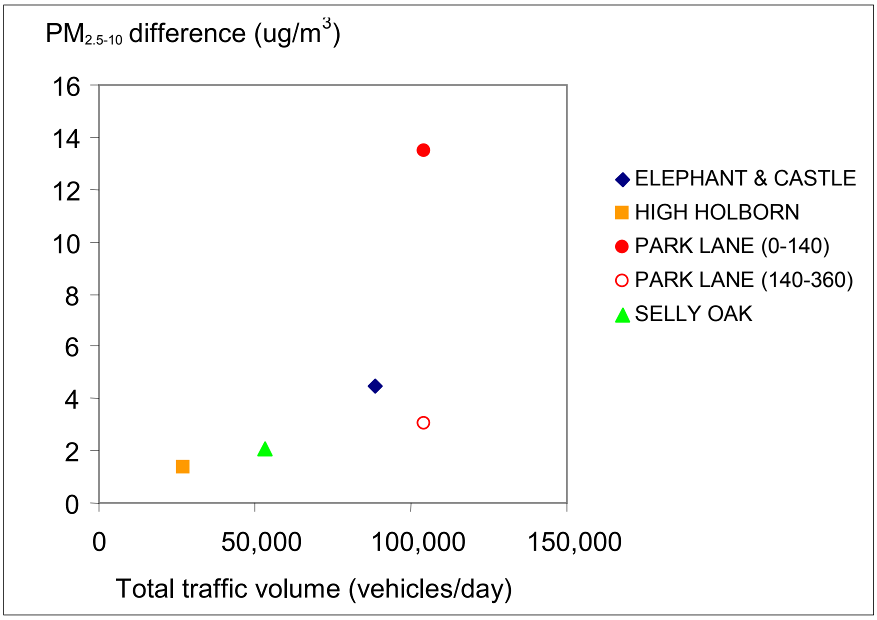

Figure 25: Roadside increment of $PM_{2.5\text{-}10}$ as a function of total traffic volume.

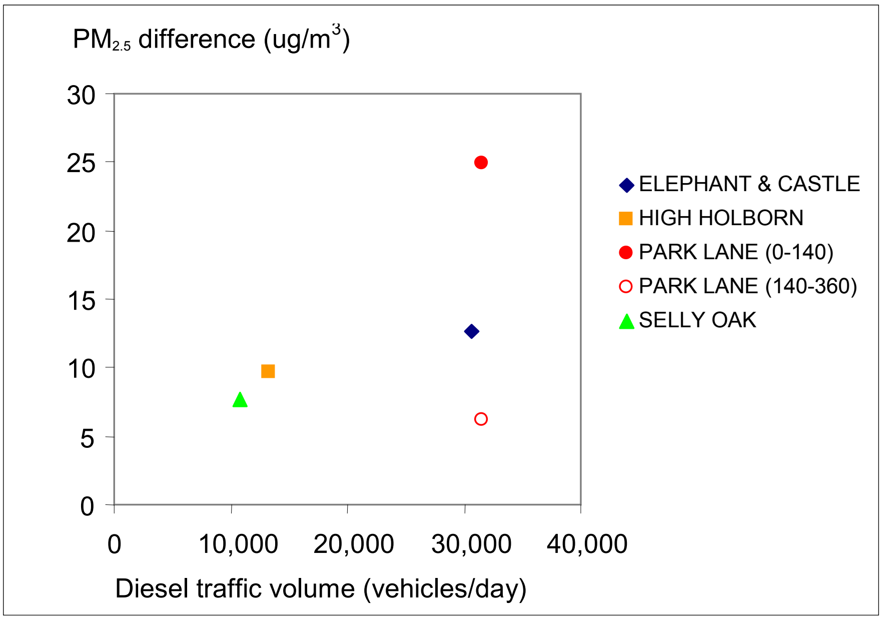

Figure 26: Roadside increment of $PM_{2.5}$ as a function of diesel traffic volume.

In order to allow for differing dispersion conditions at the various sites, the ratio of $PM_{2.5\text{-}10}$ incremental concentration to the $PM_{2.5}$ incremental concentration was also calculated. Both fine and coarse particles should be diluted in a similar way by increased wind speed. The data were also normalised to a wind speed of 7 knots (8 mph) which allows for the differing behaviour of fine and coarse particles at high wind speeds (arising from wind-driven resuspension processes for coarse particles). When this ratio was plotted against the HGV and bus traffic volume, it showed an approximately linear relationship (Figure 27). Deviations from a simply linear relationship are likely to arise from different characteristics of the sites in relation to predominant driving modes and traffic speeds, all of which influence abrasive and resuspended emissions.

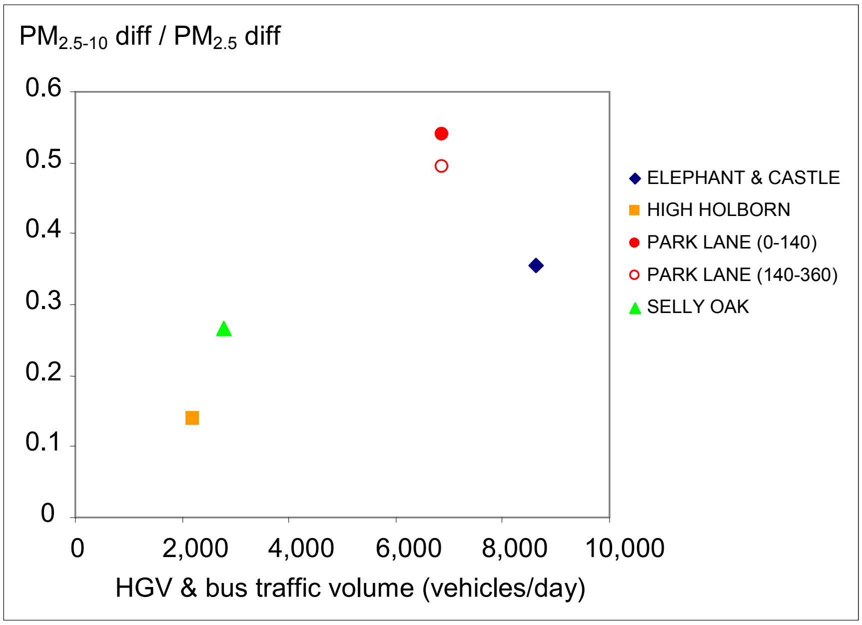

Figure 27: Ratio of traffic increment $PM_{2.5\text{-}10}$ to $PM_{2.5}$ as a function of HDV Traffic volume (concentrations normalised to a wind speed of 7 knots).

5 Model application

The final stage in the development and application of new emission factors for the estimation of non-exhaust PM emissions in the UK involved the following two steps:

(i) Identification of a final set of emission factors.

(ii) Weighting of these emission factors by traffic activity and fleet composition data to determine emissions in the UK.

5.1 Identification of emission factors

Because of the absence of significant new sets of data, no new emission factors (from a source perspective) could be developed in the project for tyre, brake and road surface wear. For these sources the EMEP modelling method was therefore used in this application phase. However, it should be noted that for tyre wear the relationship between emissions and speed in EMEP is based on very few data, and calculations performed using the HDM-4 approach produced rather different results. As the EMEP method is currently used to predict tyre and brake wear emissions in the NAEI, the results for these sources should be the same, and this was tested by calculation.

For resuspension and combined resuspension and road surface wear, the emission factors derived from the analysis of the Marylebone Road and Bloomsbury data were presented in Tables 24 and 25. Given the concerns expressed earlier about the derived emission factors, only broadly indicative values were available. For resuspension only, estimated typical values of coarse PM emissions from HDVs and LDVs were taken to be the averages of the values presented for 2000-2002. These average values were 141 mg/vkm for HDVs, and 0.8 mg/vkm for LDVs. The corresponding values for combined resuspension and road surface wear were 158 mg/vkm for HDVs, and 4.2 mg/vkm for LDVs. The difference between the emission factor for combined resuspension and road surface wear and the value for resuspension is equivalent to the PM_{coarse} emission factor for road surface wear given in EMEP (*i.e.* 17 mg/vkm for HDVs, and 3.4 mg/vkm for LDVs). It should again be noted that the resuspension emission factors should be considered to be specific to the Marylebone Road/Bloomsbury sites, and specific to the three years studied. When using these emission factors to estimate emissions due to resuspension in the UK, it is assumed that the emission factors are universally applicable. This is unlikely to be the case, and variation in factors such as speed and silt loading will probably have a substantial impact on the overall results. The resuspension emission factor for motorcycles was assumed to be zero.

5.2 Estimation of non-exhaust PM emissions in the United Kingdom

The final emission factors identified in the previous Section were weighted by the traffic activity statistics used in the NAEI in order to calculate emissions of PM_{10}, $PM_{2.5}$, PM_1 and PM_{coarse} in the UK. The activity data were supplied by NETCEN, with the total number of vkm travelled in the UK being available for the following cases:

- Six vehicle categories:
  - Cars
  - LGVs
  - Rigid HGVs
  - Articulated HGVs
  - Buses
  - Motorcycles

- Three types of road:
  - Urban roads
  - Rural roads
  - Motorways
- 39 reference years:
  - 1970-2005
  - 2010, 2015, 2025

Regional data for England, Scotland, Wales and Northern Ireland were also available for the same vehicle categories and road types, but only for the reference years 2002 and 2003.

5.2.1 Estimation of tyre, brake and road surface wear emissions

Total UK PM emissions due to tyre, brake and road surface wear, calculated using the EMEP method are shown in Appendix I, with the regional emissions for 2002 and 2003 being given in Appendix J. Consistency with the NAEI was checked. For tyre and brake wear the estimates matched those in the NAEI within a few per cent, with the differences probably being due to rounding errors. There are no estimates of PM emissions due to road surface wear in the NAEI. However, the calculations conducted using the EMEP method indicate that total UK emissions from this source could be as high as those due to tyre and brake wear (Figure 28), although the emission factors used are highly uncertain.

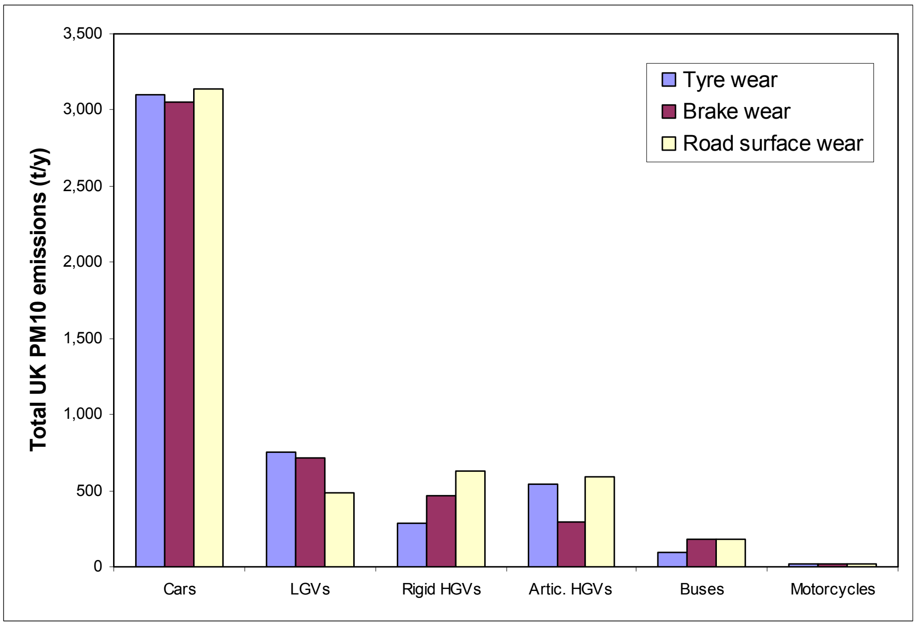

Figure 28: Total UK PM_{10} emissions due to tyre wear , brake wear and road surface wear (EMEP method).

The sensitivity of total UK emissions to the model used was also examined. The results are given in Figures 29 to 31, in which the EMEP calculations for PM_{10}, $PM_{2.5}$, PM_1 and PM_{coarse} are compared with the estimates obtained using RAINS, CEPMEIP and MOBILE 6.2 for the reference year 2005. The model used had a large influence on the predicted emissions of PM_{10}, $PM_{2.5}$ and PM_{coarse}, and differences at the emission factor level do not appear to 'cancel out' when applied at the national level.

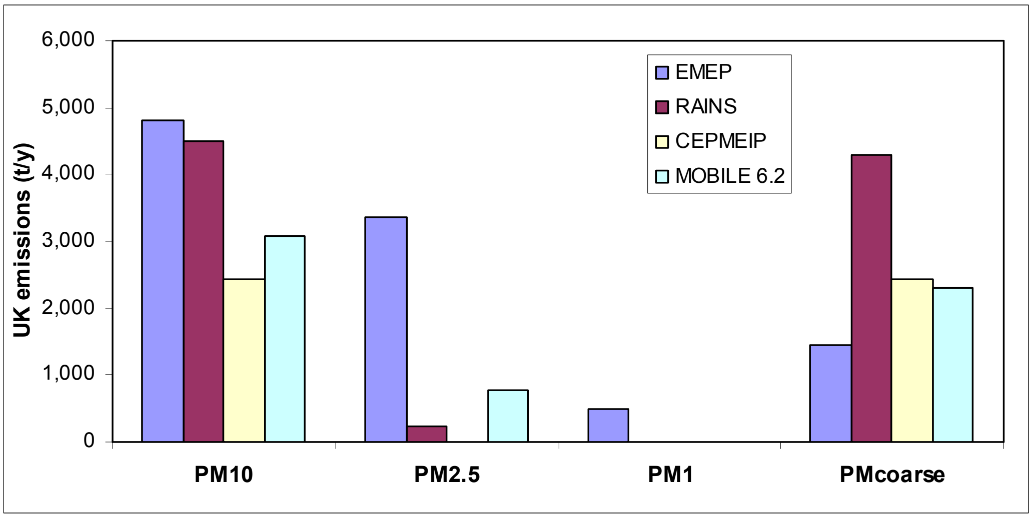

Figure 29: Total tyre wear PM emissions in the UK by model (reference year 2005).

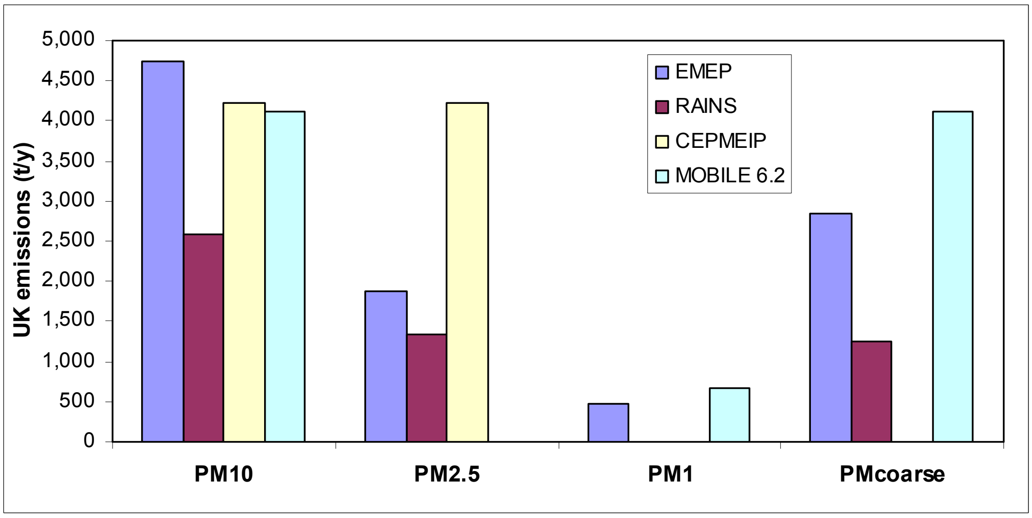

Figure 30: Total brake wear PM emissions in the UK by model (reference year 2005).

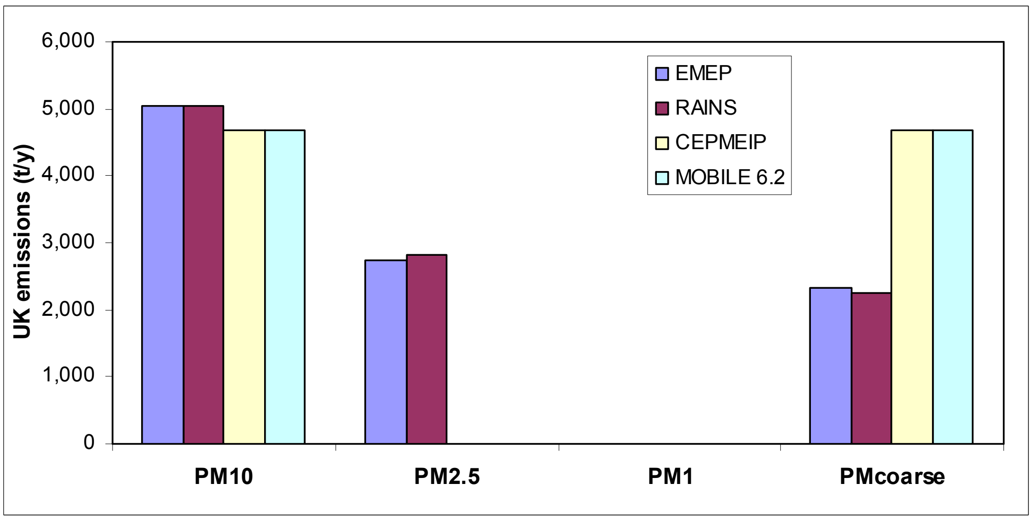

Figure 31: Total road surface wear PM emissions in the UK by model (reference year 2005).

5.2.2 Estimation of resuspension emissions

PM_{coarse} emissions due to resuspension in the UK were estimated using the emission factors described in Section 5.1. The time series of emissions for urban roads, rural roads and motorways are shown in Figures 32-34. The estimates of resuspension are directly dependent upon the amount of travel (*i.e.* vkm) on the three types of road, and in particular the amount of travel undertaken by heavy-duty vehicles. Total emissions of PMcoarse on urban roads were found to be relatively stable with time, whereas emissions on rural roads and motorways increased between 1970 and 2005, and are predicted to increase further in the future. Emissions were also calculated using the PM_{10} emission factor for resuspension of 40 mg/vkm stated in the NAEI. This value is applicable to all types of vehicle, and for the purposes of this study it was assumed to be equivalent to PM_{coarse}. Figures 35-37 show that the application of this emission factor to the UK produces substantially larger estimates of emissions than those obtained using the Marylebone emission factors. A present there is no way of knowing which of these sets of results represents the better estimate of emissions due to resuspension in the UK. The validity of different emission factors in relation to the prediction of local air pollution will be examined in Task 3 of the project.

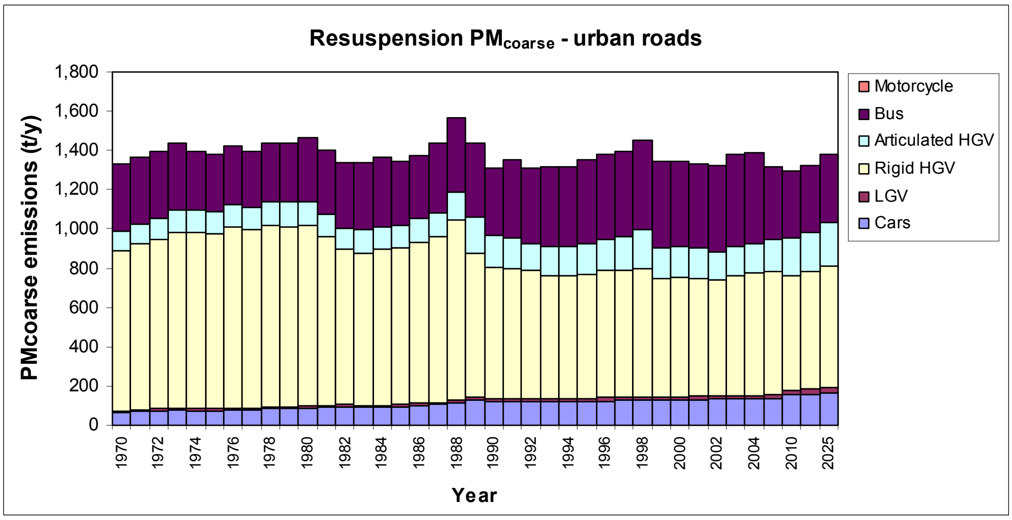

Figure 32: Total UK PM_{coarse} emissions due to resuspension – urban roads.

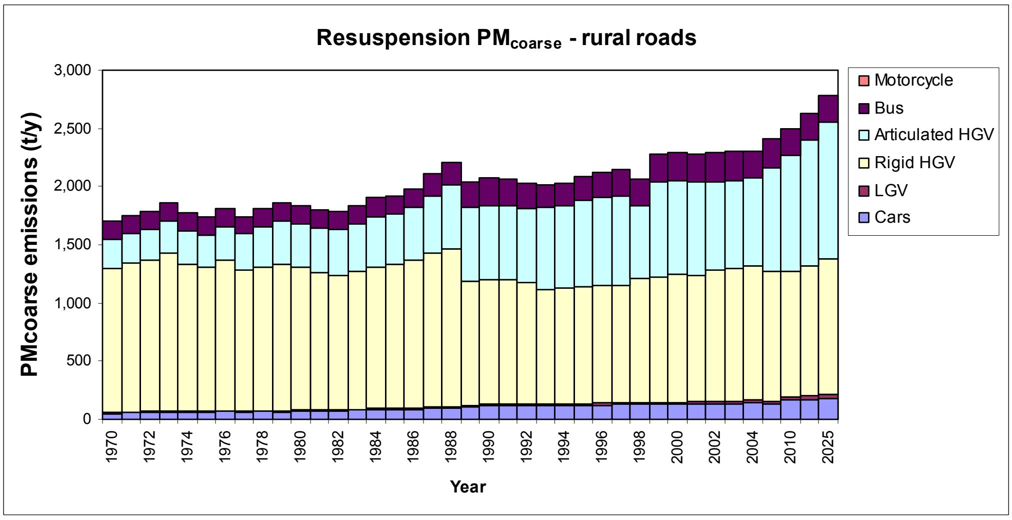

Figure 33: Total UK PM_{coarse} emissions due to resuspension – rural roads.

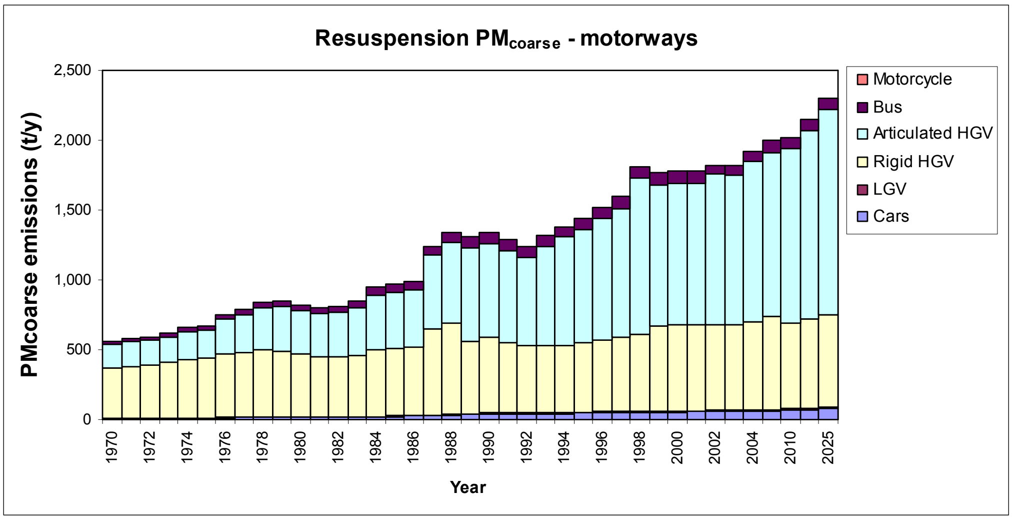

Figure 34: Total UK PM_{coarse} emissions due to resuspension – motorways.

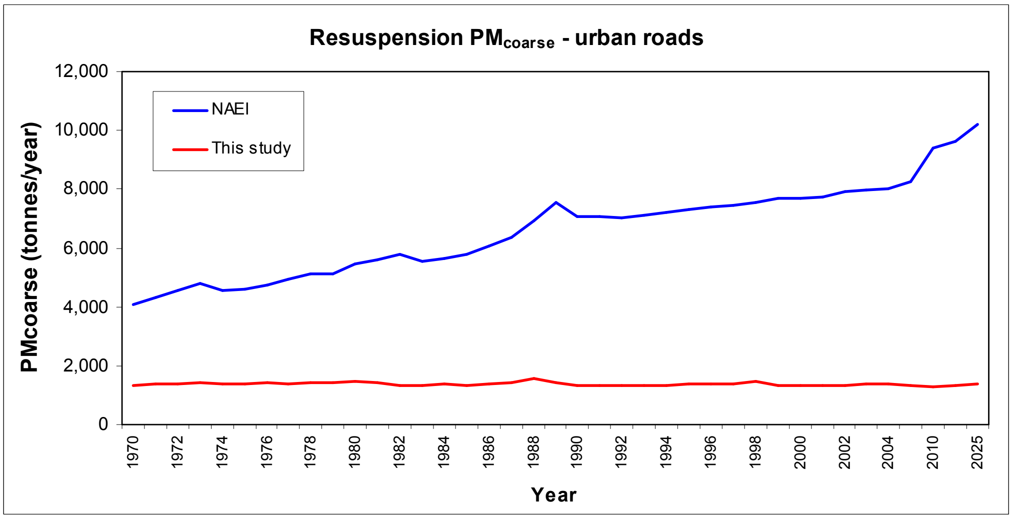

Figure 35: UK PM_{coarse} emissions due to resuspension – comparison with NAEI for urban roads.

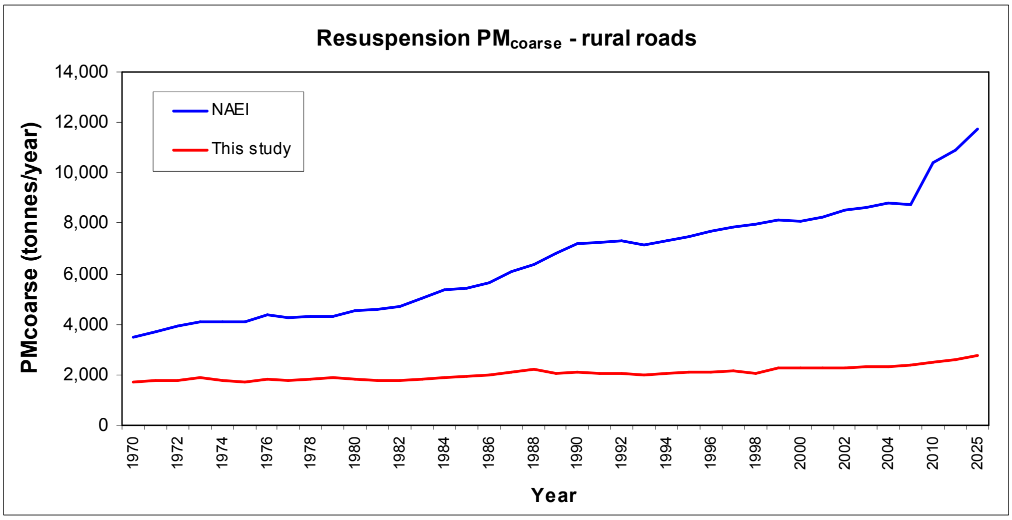

Figure 36: UK PM_{coarse} emissions due to resuspension – comparison with NAEI for rural roads.

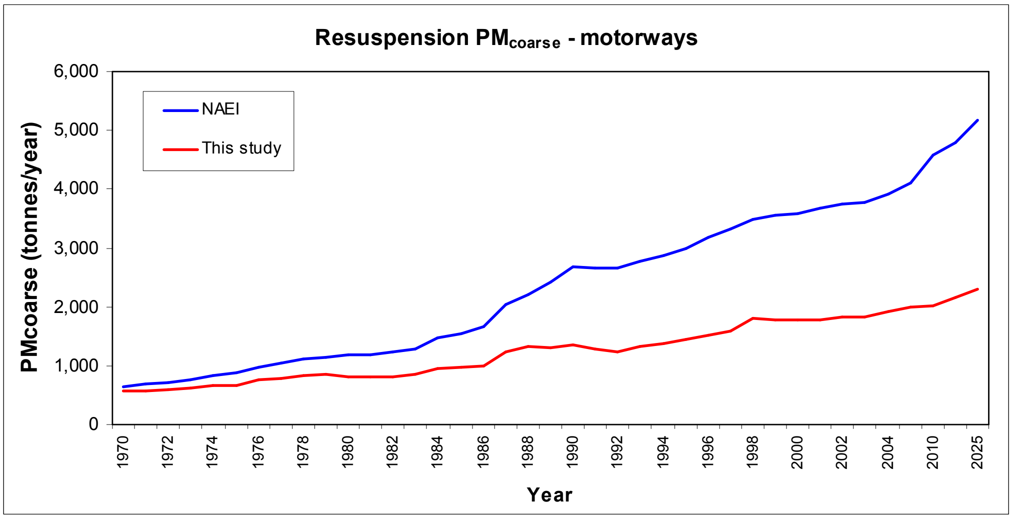

Figure 37: UK PM_{coarse} emissions due to resuspension – comparison with NAEI for motorways.

6 Summary and conclusions

6.1 Summary

This report has presented the findings of Task 2, the aim of which was to evaluate the existing models for non-exhaust PM, to develop improved modelling approaches for use in the NAEI, and to apply these models to the UK. This second task can be further broken down as follows:

Task	2a	Model evaluation and development
	2a(i)	Emissions due to abrasion sources (TRL)
	2a(ii)	Emissions due to resuspension (DEHRM)
Task	2b	Model application (TRL)

6.1.1 Model identification and comparison

The first stage of the model development process was the identification of existing models, for which a number of sources of information were considered, including the review by Boulter (2005a), further searches of publication databases and the internet, direct approach to tyre manufacturers, direct approach to brake manufacturers, and direct approach to model developers and researchers. However, other than the models described by Boulter (2005a), no other methodologies for the abrasion sources were obtained. The model evaluation for abrasion sources therefore focussed on the EMEP, RAINS, CEPMEIP and MOBILE6.2 methods identified in the review.

In the absence of independent test data it was not possible to assess the absolute accuracy of the different approaches. The evaluation was therefore based upon a between-model comparison of emission factors for different size fractions and different vehicle categories. For most vehicle categories, the different models/databases produce substantially different emission factors. The EMEP method, which is currently used in the NAEI, is the most detailed approach, incorporating corrections for both speed and, in the case of HDVs, vehicle load. For tyre wear, different emission factors are also provided in EMEP for different types of HDV, based on the number of axles. It was concluded that none of the other three models or databases considered in the evaluation would currently offer any advantage over the EMEP method for application in the UK.

The approaches to tyre and brake manufacturers, to other research institutions, and to model developers yielded little information which could be used directly in the project. However, one other modelling approach was considered to be directly relevant to this project - the mechanistic model used for predicting tyre wear in HDM-4.

6.1.2 Abrasion model development

The abrasion model development phase of the work proceeded along the following lines:

- Further development of the EMEP method.
- Adaptation of the tyre wear model in HDM-4.
- Use of brake wear data from the US relating to PM emissions per braking event.

The lack of a substantial amount of new source-specific emission factors in the literature meant that no developments of the EMEP method were actually possible. Similarly, at present, brake wear data only exist for a limited number of braking conditions, and therefore the development of a prediction method according to this approach has not yet been possible. However, a spreadsheet-based version of the tyre consumption method in HDM-4 was compiled during the project, and a simple module was added to enable the prediction of PM_{10} emissions due to tyre wear. An attempt was also made to assess the applicability of the prediction methods for total non-exhaust particles to UK conditions.

Five separate models were eventually carried forward into Task 2a(ii), the determination of emission factors for resuspension. These models were:

- The existing EMEP method.
- The RAINS database.
- The CEPMEIP database.
- The HDM-4 model for tyre wear (combined with a PM_{10} emission calculation routine).
- The German traffic situation model.

6.1.3 Resuspension model development

In Task 2a(ii), three main emission modelling methods (EMEP, RAINS, CEPMEIP) were used by DEHRM to develop an improved method for estimating emissions due to resuspension at particular localities, and specifically for Marylebone Road in London. Although it was also concluded from Task 2a(i) that neither RAINS nor CEPMEIP would currently offer any advantage over the EMEP method for application in the UK, the RAINS and CEPMEIP databases were used to provide an indication of the sensitivity of the abrasive source estimate to the model/database used. The HDM-4 tyre wear model was also used in Task 2a(ii) in order to assess the effects of applying a detailed model to calculate tyre wear emissions. Although there are concerns about the representativeness of the German traffic situation model for the UK, it was also considered appropriate to include this model for further sensitivity testing.

In order to estimate PM_{10} emissions due to resuspension - Task 2(ii) - DEHRM applied a method based upon simultaneous measurement of PM_{10} and $PM_{2.5}$ in ambient air at roadside and background monitoring sites. The total emissions attributable to non-exhaust sources were expressed as the sum of the contribution of abrasion sources (brake, tyre, and road wear) and the resuspended component:

$$\boldsymbol{E_{TOTAL} = E_{BRAKE} + E_{TYRE} + E_{ROAD} + E_{RESUSP}}$$

The above equation was rearranged to obtain an estimate of resuspension. Emissions for brake, tyre, and road wear were estimated from the aforementioned emission factors and traffic fleet data for 2000, 2001 and 2002. Total PM_{10} emissions from the traffic were calculated on the basis of measured roadside increments of PM_{10} and NO_x, and calculated NO_x emissions. It was assumed that the roadside increment $PM_{2.5}$ was solely attributable to vehicular exhaust sources and that non-exhaust emissions were largely confined to the $PM_{2.5\text{-}10}$ size fraction. NO_x emission factors (g/km) were calculated by applying the polynomial equations given by the NAEI

Since EMEP provides emission factors for a greater range of vehicle categories, and includes corrections for vehicle speed and heavy goods vehicle load, the use of the abrasion emissions derived using this method were favoured in the calculation of resuspension. Comparison of the estimated resuspended component with total non-exhaust emissions suggested that resuspension accounts for 43-49% of the non-exhaust emissions. Furthermore, resuspension emissions were found to be around 30% of the magnitude of exhaust emissions.

HDVs were found to be almost entirely responsible for the resuspension of particles. Emission factors of $PM_{2.5\text{-}10}$ from HDVs due to resuspension ranged from 139 mg/vkm to 145 mg/vkm. These values appear to be lower than those reported for other countries and conditions in the literature.

Much smaller emission factors were obtained for LDVs. Indeed, a negative value was obtained for 2002, although the value was within the uncertainty of the calculation.

However, there was found to be a decrease in emissions of resuspended particles in the $PM_{2.5\text{-}10}$ size range between 2000 and 2002. This may have arisen from the use of a constant $PM_{2.5\text{-}10}$:PM_{10} emissions ratio of 0.4. If the observed temporal decrease (2000-2002) calculated for PM_{10} emissions were the result of a reduction in exhaust emissions of PM, then a change in the $PM_{2.5}$:PM_{10} ratio would be anticipated, assuming no change in non-exhaust emissions. The appropriateness of applying a constant ratio was therefore investigated using a variable $PM_{2.5\text{-}10}$:PM_{10} ratio to calculate total coarse particle emissions and to derive estimates of resuspension. However, the use of observed $PM_{2.5\text{-}10}$:PM_{10} ratios to calculate $PM_{2.5\text{-}10}$ emissions, and subsequently to estimate resuspension, offers little improvement over the use of the constant ratio.

Due to the unexpected decreasing trend in the $PM_{2.5\text{-}10}$: PM_{10} ratio, a further investigation was undertaken of the trends in the measured concentrations of the three PM metrics at the Marylebone Road and Bloomsbury sites, and the trend in the roadside increment.

In contrast to the anticipated increase in the $PM_{2.5\text{-}10}$:PM_{10} ratio with time, due to the reduction in exhaust PM emissions, there was no systematic trend in the observed ratio for Marylebone Road. Over the time period used in the estimation of resuspension emissions the ratio showed a tendency to decrease. For the Bloomsbury site the ratio remained rather static from 1998 to the end of 2000, with a typical value of 0.35. In the latter half of 2001 and throughout 2002 the ratio increases markedly to around 0.50, indicative of construction activity close to the monitoring site resulting in the emission of coarse PM. The ratio reduced gradually during 2003 to a steady level of 0.38 – 0.40. The decreasing trend of the proportion of PM_{10} composed of coarse PM in the Marylebone Road data was accentuated in the roadside increment. From a maximum value of greater than 0.50 around September 1999, the ratio fell substantially to less than 0.25 by the end of 2001. This is a reflection of the large increase in $PM_{2.5\text{-}10}$:PM_{10} ratio at Bloomsbury throughout 2001 particularly, and highlights that during these months the use of Bloomsbury as a background site by which to calculate roadside increments may be questionable. The trend reversed in 2002, and the proportion of coarse PM appeared to rise continually to a value in excess of 0.55. It can be concluded from the observations discussed above that the anticipated increase in the ratio $PM_{2.5\text{-}10}$:PM_{10} is not conclusively supported by data from Marylebone Road or the roadside increment data. It is also apparent that the selection of appropriate background sites for calculation of roadside incremental concentrations is imperative. The effect of local sources upon concentrations measured at London Bloomsbury may go some way to explaining the unexpected trends in the proportion PM_{10} composed of coarse PM in the roadside increment at Marylebone Road.

There was evidence of an increase in resuspension with wind speed, with the rate of increase apparently slowing as wind speeds get larger. However, the apparently large wind speed dependence of resuspension leaves the question of how there is apparently such a strong dependence on heavy-duty vehicles but not light-duty vehicles. It was postulated that the key role of the vehicle is in resuspending the particulate matter from the road surface initially, be it by tyre shear or vehicle-generated turbulence. The role of wind speed is in generating the more extensive atmospheric turbulence responsible for keeping the larger particles resuspended such that they then have a significant influence upon airborne concentrations.

Analysis of the Marylebone Road data revealed no clear relationship between resuspension and precipitation. It appears that simply looking at rainfall totals against resuspension is inadequate for the task of deriving a relationship between the two.

For Marylebone Road, the emissions calculated obtained using the HDM-4 tyre wear model were compared with those calculated using the EMEP method. The results indicated that the HDM-4 method yields daily tyre wear emissions that were typically half of the values produced by the EMEP method. The calculations conducted using the HDM-4 model showed that emissions from two-wheel motor vehicles are of greater importance than in EMEP. The tyres of motorcycles are known to wear at a relatively high rate, and this is taken into account in HDM-4. However, in this project a single PM_{10} proportion of tyre wear was applied to all vehicles in HDM-4, but it is not known whether this is appropriate. For example, whilst the tyre wear rate for two-wheel vehicles is high, the PM_{10} proportion may be low, and this requires further investigation.

Separate calculations conducted using the German traffic situation model at Marylebone Road showed that the model produces values that are too large, indeed in excess of the particle source strength itself. This is likely to be a consequence of applying emission factors calculated from research on German roads to a road in a different environment, and the need to define different levels of congestion arbitrarily. Since the methodology behind the traffic situation approach is similar to ours, in obtaining relationships between NO_x and PM_{10} based on atmospheric measurements, revision of the emission factors in the traffic situation model for use in the UK could provide a useful tool for deriving estimates of non-exhaust PM emissions.

A more constrained analysis was conducted using the data from other air pollution monitoring sites in London and Birmingham, primarily to investigate its consistency with the conclusions drawn from Marylebone Road. From the results obtained at Marylebone Road it would be anticipated that the roadside increment of $PM_{2.5\text{-}10}$ would scale according to the volume of heavy-duty vehicles. At the other sites there was a general upward trend for coarse particle incremental concentration with total traffic, and $PM_{2.5}$ showed a relatively linear increase with diesel traffic volume. In order to allow for differing dispersion conditions at the various sites, the ratio of $PM_{2.5\text{-}10}$ incremental concentration to the $PM_{2.5}$ incremental concentration was also calculated. The data were also normalised to a wind speed of 7 knots (8 mph) which allowed for the differing behaviour of fine and coarse particles at high wind speeds. When this ratio was plotted against the HGV and bus traffic volume, it showed an approximately linear relationship. Deviations from a simply linear relationship are likely to arise from different characteristics of the sites in relation to predominant driving modes and traffic speeds, all of which influence abrasive and resuspended emissions.

6.1.4 Model application

The final stage in the development and application of new emission factors for the estimation of non-exhaust PM emissions in the UK involved the identification of a final set of emission factors, and the weighting of these emission factors by traffic activity and fleet composition data to determine emissions in the UK.

Because of the absence of significant new sets of data, no new emission factors could be developed in the project for tyre, brake and road surface wear. For these sources the EMEP modelling method was therefore used in this application phase. However, it should be noted that for tyre wear the relationship between emissions and speed in EMEP is based on very few data, and calculations performed using the HDM-4 approach produced rather different results. As the EMEP method is currently used to predict tyre and brake wear emissions in the NAEI, the results for these sources should be the same, and this was tested by calculation.

For resuspension only broadly indicative values were available from the analysis of the Marylebone Road and Bloomsbury data. For resuspension only, estimated typical values of coarse PM emissions from HDVs and LDVs were taken to be the averages of the values presented for 2000-2002. These average values were 141 mg/vkm for HDVs, and 0.8 mg/vkm for LDVs. The emission factors for motorcycles associated with resuspension and road surface wear were assumed to be zero. These emission factors were used to estimate UK emissions of PM_{coarse} due to resuspension. This necessarily assumes that the emission factors are universally applicable. This is unlikely to be the case, and variation in factors such as speed and silt loading will probably have a substantial impact on the overall results. The emission factors should really be considered to be specific to the Marylebone Road/Bloomsbury sites, and specific to the three years studied.

The final emission factors identified in the previous Section were weighted by the traffic activity statistics used in the NAEI in order to calculate emissions of PM_{10}, $PM_{2.5}$, PM_1 and PM_{coarse} in the UK. The activity data were supplied by NETCEN, and the statistics took the following form:

The total number of vkm travelled in the UK were available for the following cases:

- Six vehicle categories:
  - Cars
  - LGVs
  - Rigid HGVs
  - Articulated HGVs
  - Buses
  - Motorcycles
- Three types of road:
  - Urban roads
  - Rural roads
  - Motorways
- 39 reference years:
  - 1970-2005
  - 2010, 2015, 2025

Regional data for England, Scotland, Wales and Northern Ireland were also available for the same vehicle categories and road types, but only for the reference years 2002 and 2003.

Total UK PM emissions due to tyre, brake and road surface wear, calculated using the EMEP method are shown in Appendix I, with the regional emissions for 2002 and 2003 being given in Appendix J. Consistency with the NAEI was checked. For tyre and brake wear the estimates matched those in the NAEI within a few per cent, with the differences probably being due to rounding errors. There are no estimates of PM emissions due to road surface wear in the NAEI. However, the calculations conducted using the EMEP method indicate that total UK emissions from this source could be as high as those due to tyre and brake wear, although the emission factors used are highly uncertain.

The sensitivity of total UK emissions to the model used was also examined. The results are given in Figures 29 to 31, in which the EMEP calculations for PM_{10}, $PM_{2.5}$, PM_1 and PM_{coarse} are compared with the estimates obtained using RAINS, CEPMEIP and MOBILE 6.2 for the reference year 2005. The model used had a large influence on the predicted emissions of PM_{10}, $PM_{2.5}$ and PM_{coarse}.

PM_{coarse} emissions due to resuspension in the UK were estimated using the emission factors described in Section 5.1. Total emissions of PMcoarse on urban roads were found to be relatively stable with time, whereas emissions on rural roads and motorways increased between 1970 and 2005, and are predicted to increase further in the future. The estimates of resuspension are directly dependent upon the amount of travel (*i.e.* vkm) on the three types of road, and in particular the amount of travel undertaken by heavy-duty vehicles. Emissions were also calculated using the PM_{10} emission factor for resuspension of 40 mg/vkm stated in the NAEI. This value is applicable to all types of vehicle, and for the purposes of this study it is assumed to be equivalent to PM_{coarse}. The application of this emission factor to the UK produces substantially larger estimates of emissions than those obtained using the Marylebone emission factors. A present there is no way of knowing which of these sets of results represents the better estimate of emissions due to resuspension in the UK. The validity of different emission factors in relation to the prediction of local air pollution will be examined in Task 3 of the project.

6.2 Conclusions

The following conclusions were drawn from this work:

(i) For most vehicle categories, the different models/databases for PM due to abrasion produce substantially different emission factors, but in the absence of independent test data it is not possible to assess the absolute accuracy of the different approaches.

(ii) The EMEP method, which is currently used in the NAEI, is the most detailed approach, incorporating corrections for both speed and, in the case of HDVs, load.

(iii) Given that there have been few new measurements and methodological developments for the abrasion sources in recent years, further refinement to the EMEP method is not currently possible.

(iv) The HDM-4 tyre consumption model offers possibilities in terms of modelling PM emissions, but until further work is performed to develop suitable parameters to apply the full theoretical model, the predictions from the HDM-4 model should be viewed with caution.

(v) At present, brake wear PM data only exist for a limited number of braking conditions, and therefore the development of a prediction method which combines emission factors for different types of stop with driving pattern data has not yet been possible.

(vi) For Marylebone Road, comparison of the estimated resuspended PM component with total non-exhaust PM emissions suggested that resuspension accounts for 43-49% of the non-exhaust emissions. Furthermore, resuspension emissions were found to be around 30% of the magnitude of exhaust emissions.

(vii) HDVs were found to be almost entirely responsible for the resuspension of particles. Emission factors for resuspension due to HDVs ranged from 139 mg/vkm to 145 mg/vkm. These values appear to be lower than those reported in the literature. Much smaller emission factors were obtained for LDVs.

(viii) The separation of weekday and weekend data, the latter influenced much less than the former by heavy-duty vehicles, clearly indicates that it is the heavy-duty traffic which plays far the major role in driving resuspension processes.

(ix) There was found to be a decrease in emissions of resuspended particles in the $PM_{2.5-10}$ size range between 2000 and 2002, due in part to the use of a constant $PM_{2.5-10}$:PM_{10} emissions ratio of 0.4. However, the use of observed $PM_{2.5-10}$:PM_{10} ratios to calculate $PM_{2.5-10}$ emissions, and subsequently to estimate resuspension, offers little improvement over the use of the constant ratio. Furthermore, the anticipated increase in the ratio $PM_{2.5-10}$:PM_{10} is not conclusively supported by data from Marylebone Road or the roadside increment data. It is also apparent that the selection of appropriate background sites for calculation of roadside incremental concentrations is imperative.

(x) Inevitably there are significant uncertainties surrounding both the calculated NO_x emissions and, to a greater extent, the estimates of abrasion emissions which appear quite sensitive to the method adopted. Nonetheless, the calculated values of resuspension appear to be of a reasonable magnitude, and their variation with wind speed is broadly consistent with that which would be anticipated.

(xi) It is postulated that the key role of the vehicle is in resuspending the particulate matter from the road surface initially, be it by tyre shear or vehicle-generated turbulence. The role of wind speed is in generating the more extensive atmospheric turbulence responsible for keeping the larger particles resuspended such that they then have a significant influence upon airborne concentrations.

(xii) Whilst the data from the TRAMAQ UG250 project are far less suited to detailed analysis than those from Marylebone Road, close examination does demonstrate that they are broadly consistent with predictions drawn from the Marylebone Road conclusions.

6.3 Recommendations

This study has shown that there are few detailed methodologies for predicting emissions of particulate matter from non-exhaust sources. Furthermore, there has been insufficient information presented in the recent literature to enable the further development of existing models from a source perspective, and approaches to tyre and brake manufacturers, to other research institutions, and to model developers did not prove to be productive. There is clearly a need for more extensive empirical data for use in emission models.

It also clear from this study that significant quantitative insights into non-exhaust PM can be gained from analysis of measured data from a heavily instrumented monitoring site, such as the one at Marylebone Road. Analysis of less comprehensive datasets can be useful for qualitative confirmation of conclusions, but are unlikely to yield genuinely quantitative results. However, one weakness of the resuspension estimates in current study is that they are based purely on a single street canyon site, and it is important to test the extent to which the results can be generalised to other locations having different characteristics. It should also be recognised that the results for emission factors for resuspension are critically dependent upon the estimates of abrasive emissions which, as shown in this work, can vary significantly between different models used for their estimation. Consequently, in any study in which resuspended PM is determined as the difference between total non-exhaust PM and abrasion-related PM, reliable estimates of abrasive emissions are crucial.

6.3.1 Recommendations for Task 3

Task 3 of the project involves the development and application of a modelling methodology to estimate the contribution of different sources of non-exhaust PM to ambient concentrations in the UK. The model (ADMS-Urban) will be used to predict the contributions to ambient PM_{10} and $PM_{2.5}$ concentrations of tyre wear, brake wear and road surface wear for cars, light goods vehicles and heavy-duty vehicles. Further disaggregation will be made where possible.

The emission factors generated in Task 2 will be used in Task 3, and in this respect the emphasis should be on the EMEP method and the resuspension emission factors from Marylebone Road. The re-calculated sources of non-exhaust particulate matter will be entered into the model. PM concentration estimates will be made for a selection of representative roadside and background locations in the UK. Clearly, it is important that the model is applied to the sites used to estimate resuspension in Task 2, notably Marylebone Road, although the model should also be applied to data sets which are independent of those used to derive emission factors.

6.3.2 Recommendations for future work

Some specific recommendations for future investigations, based on the work conducted so far on the project, are as follows:

(i) *Assessment of tyre, brake and road surface materials in use*. Many different brake and tyre types are in use in the UK, but little information on the structure of the UK market and the performance of different materials has been reported in the scientific literature. Similarly, there is a need to understand the different road surfaces used in the UK by road type and geographic location. A survey of materials in use (including trends in low-noise surfaces) would assist in the design of future experimental work and modelling approaches.

(ii) *Determination of component wear factors*. Although some wear factors for tyres and brake linings are available in the literature, there is little information specifically for the UK, and the relationships between wear and real-world vehicle operation are not well understood. Furthermore, the relative wear factors for brake linings and brake rotors/drums are not well documented. Further information of this type is required for modelling purposes. An appropriate study could also include an assessment of tyre pressure and tracking for in-service vehicles, and corresponding effects on component wear.

(iii) *Compositional profiles of source materials*. The compositions of the various tyre and brake lining formulations used in the UK have not been reported in detail. An extensive examination of the composition of the tyres and brakes available on the UK market is required. It would also be useful to examine the extent to which tyre tread and brake lining material are altered during use, and to analyse the tyres, the brake lining, and the brake dust from same in-service vehicles at regular intervals.

(iv) *Laboratory-based characterisation of emissions*. Further sampling of tyre, brake and road surface wear particles is required under controlled laboratory conditions, and using analytical equipment which cannot easily be deployed in the field. Ideally, particle size distributions should be determined with a high resolution, and chemical composition measurements of the emitted PM should be conducted to complement (iii).

(v) *Real-world measurements using instrumented vehicles*. The measurement of non-exhaust particles could be conducted *in situ* under a range of real-world vehicle operating conditions. Some experiments of this type have already been conducted in the United States.

(vi) *New receptor modelling studies*. Receptor modelling studies at a varied range of locations, and based on new source profiles, would contribute significantly to the understanding of non-exhaust PM. Ideally, something of the order of five or six site pairs (roadside and urban background) would be studied in order to obtain insights into the consistency of findings and the ranges of behaviour. Again, particle size distributions should be determined in the roadside and background atmosphere, with high size resolution in order to define more clearly whether there is a clear size cut between the exhaust and non-exhaust emissions, and if not, the extent of overlap of the size distributions. Chemical composition measurements on airborne PM may further assist in distinguishing particles from different sources. Comparisons between receptor modelling studies inside and outside of road tunnels could assist in the understanding of resuspension processes.

(vii) *Investigation of wind-related effects*. One of the interesting facets of the results of this study is the wind speed dependence of the calculated resuspended particle emission factor. This is indicative of a major influence of wind speed upon road surface particle resuspension and warrants further investigation with a view to identifying and quantifying separately the role of the wind and of traffic in resuspending particles initially and in maintaining them in suspension in the atmosphere. This is an aspect which could be integrated within a study of further site pairs and would best be conducted using wind speed measurements both from above the building canopies and within the street canyon environment.

(viii) *Investigation of precipitation-related effects.* It appears that simply looking at rainfall totals against resuspension is inadequate for the task of deriving a relationship between the two. Any study of the effects of precipitation should incorporate descriptions of the duration of rainfall events, the rainfall intensity (and hence run-off, which is likely to reduce the dust available for resuspension), and evaporation, as in the model presented by Omstedt *et al*. (2005).

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Appendix A. Glossary

Accumulation mode	Particles formed via the coagulation of nucleation mode particles, primary emission sources, and gas-to-particle transformations. Particles range between around 0.05µm and 1 µm in diameter, and have an atmospheric residence time of tens of days.
Coarse particle mode	Particles larger than around 1 µm, including wind-blown crustal matter and material released during abrasion processes. Coarse particles have shorter residence times than accumulation mode particles. This is not consistent with the definition for PM_{COARSE} given above.
Dustfall	Particles larger than 100µm, which tend to fall out of the atmosphere within minutes.
HDVs	Heavy-duty vehicles (heavy goods vehicles and buses) >3.5 tonnes gross vehicle weight.
HGVs	Heavy goods vehicles.
LDVs	Light-duty vehicles (cars and light goods vehicles).
LGVs	Light goods vehicles between 2.5 tonnes and 3.5 tonnes gross vehicle weight.
NAEI	National Atmospheric Emissions Inventory.
Nucleation mode	Particles emitted directly from combustion sources, having a diameter of less than around 50nm and an atmospheric residence time of a few hours. They are transformed by coalescence and condensation into larger accumulation mode particles.
PM_{10}	Mass concentration of particles passing through a size-selective inlet designed to exclude particles greater than 10 µm aerodynamic diameter.
$PM_{2.5}$	Mass concentration of particles passing through a size-selective inlet designed to exclude particles greater than 2.5 µm aerodynamic diameter. These are sometimes referred to as 'fine' particles.
$PM_{2.5-10}$ or PM_{COARSE}	Mass concentration of 'coarse' particles, determined as the difference between PM_{10} and $PM_{2.5}$.
PM_1	Mass concentration of particles passing through a size-selective inlet designed to exclude particles greater than 1 µm aerodynamic diameter.
$PM_{0.1}$	Mass concentration of particles of diameter smaller than 0.1 µm. These are sometimes referred to as 'ultrafine' particles.
Primary particles	Particles emitted directly to the atmosphere.
Secondary particles	Particles formed within the atmosphere from gas phase precursors. This includes particles originating from atmospheric oxidation of sulphur and nitrogen oxides, and their reaction products with ammonia, and from the oxidation of organic compounds.
TSP	Total suspended particulate.

Appendix B. EMEP/CORINAIR method

B.1 Simple methodology for PM_{10}

Equation B1 is presented to calculate PM_{10} emissions from brake wear and tyre wear emissions from a given vehicle category, spatial area and time period by selecting appropriate values for the fleet size and the activity rate (mileage). Total traffic-generated emissions can be estimated by summating the emissions from individual vehicle classes.

$$E_{s,j} = N_j \times M_j \times e_{s,j} \qquad \text{(Equation B1)}$$

Where:

$E_{s,j}$ = Total PM_{10} Emissions (g) for the defined time period and spatial boundary
N_j = Number of vehicles in defined class within the defined spatial boundary
M_j = Average mileage driven (km) per vehicle in defined class during the defined time period
$e_{s,j}$ = Mass emission factor (g/km)
s = Non-exhaust emission source (tyre, brake, road surface)
j = vehicle category (two-wheel vehicle, passenger car, LGV, HDV)

The user of this simple methodology need only determine activity rates in the form of the vehicle population (by class) and mileage driven per vehicle (by class) for the requested temporal and spatial resolution. The relevant emission factors for use in Equation B1 are given in Table B1.

Table B1: Non-exhaust PM_{10} emission factors to be used with the simple methodology and comparison with aggregated exhaust emission factors.

Vehicle class (j)	Particle source and emission factor (g/km)		
	Tyre wear	Brake wear	Road surface wear
Two-wheel vehicles	0.0028	0.0037	0.0030
Cars	0.0064	0.0073	0.0075
LGVs	0.0101	0.0115	0.0075
HDVs	0.0270	0.0320	0.0380

B.2 Detailed methodology

Tyre wear particle emissions

In order to estimate particle emissions from tyre wear, Equation B2 can be used. This equation refers to a single vehicle category for a defined temporal and spatial resolution. Also, different particle size classes are considered.

$$E_{TYRE,i,j} = N_j \times M_j \times e_{TYRE,\,TSP,j} \times f_{TYRE,i} \times S_{T(V)} \qquad \text{(Equation B2)}$$

Where:

$E_{TYRE,i,j}$ = Total emissions (g) for the defined time period and spatial boundary
N_j = Number of vehicles in the defined class within the defined spatial boundary
M_j = Mileage driven (km) by vehicles in the defined class during the defined time period

$e_{TYRE,TSP,j}$ = TSP mass emission factor from tyre wear (g/km)
$f_{TYRE,i}$ = Mass fraction of tyre-wear TSP that can be attributed to particle size class ***i***
$S_{T(V)}$ = Tyre-wear correction factor for a mean vehicle travelling speed ***V***
i = Size fraction (TSP, PM_{10}, $PM_{2.5}$, PM_1 and $PM_{0.1}$)
j = vehicle category (two-wheel vehicle, passenger car, LGV, HDV)

TSP emission factors for different vehicle classes are given in Table B2. All emission factors are based on available experimental data. It should be noted that the TSP emission rates do not assume that all tyre wear material is transformed into suspended particulate, as a large fraction of tyre rubber may be produced as dustfall particles or larger shreds (*e.g.* under heavy braking). A value of 0.6 has been selected as the PM_{10}/TSP ratio for tyre wear in order to derived TSP values where PM_{10} emission rates are available in the literature.

Table B2: TSP emission factors from tyre wear.

Vehicle class (j)	Emission factor (g/km) $e_{TYRE,TSP}$
Two-wheel vehicles	0.0046
Cars	0.0107
LGV	0.0169
HDV	Equation B3

For the HDV case, emission factor needs to take vehicle size into account as follows:

$$(e_{TYRE})_{HDV} = \frac{N_{axle}}{2} \cdot LCF_{TYRE} \cdot (e_{TYRE})_{PC} \qquad \text{(Equation B3)}$$

Where:

N_{axle} = Number of truck axles
LCF_{TYRE} = Load correction factor
$(e_{TYRE})_{PC}$ = The TSP emission factor for cars

For HDVs, the number of axles is a parameter which can be used to differentiate vehicle size. An additional parameter is a load correction factor, which accounts for the load carried by the truck or bus. The load correction factor can be estimated on the basis of Equation B4 which has been derived by linear regression on experimental data:

$$LCF_{TYRE} = 1.41 + (1.38 \times LF) \qquad \text{(Equation B4)}$$

Where ***LF*** is the load factor for the truck, ranging from 0 for an empty truck to 1 for a fully laden one. The same equations can be used for urban buses and coaches.

Typical size profiles for TSP emitted by tyre wear have been obtained by combining information from the literature. Based on this information, the mass fraction of TSP in the different particle size classes is shown in Table B3.

Table B3: Size distribution of tyre wear emitted particles.

Particle size class (i)	Mass fraction (f_{TYRE}) of TSP
TSP	1.000
PM_{10}	0.600
$PM_{2.5}$	0.420
PM_1	0.060
$PM_{0.1}$	0.048

A speed correction is required to account for the different wear factor of the tyre depending on the vehicle speed. Figure B1 shows the speed correction, based on the findings of Luhana *et al*. (2002). It should be noted that, as in the case of exhaust emission factors, vehicle speed corresponds to mean trip speed and not constant travelling speed. There is a decreasing pattern of emissions with increasing speed. This is in contrast to the usual perception that airborne particles increase in the wake of a vehicle as speed increases, because Figure B1 corresponds to primary particle emissions from the tyre and not resuspended dust. Tyre wear decreases as mean trip speed increases, because braking and cornering are more frequent in urban driving than in motorway driving. Note that $S_{T(V)}$ = 1 when the mean trip speed is 80 km/h, and stabilises below 40 km/h and above 90 km/h due to the absence of any experimental data. Also, although the proposed equation has been obtained from measurements on passenger cars, it is to be used for all vehicle categories.

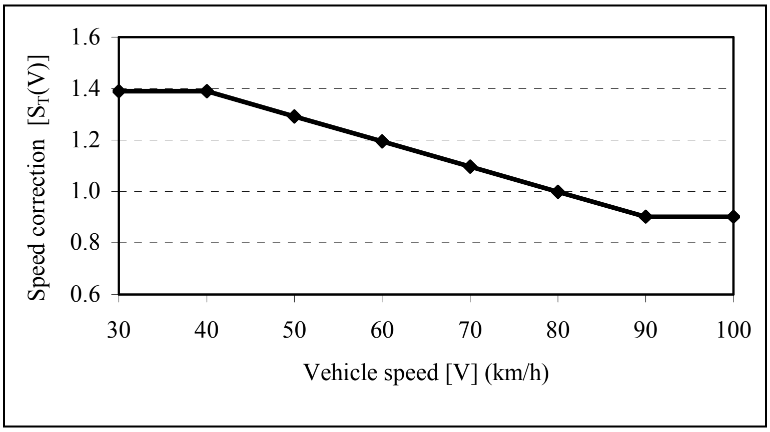

Figure B1: Speed correction factor($S_{T(V)}$) for tyre wear particle emissions.

Brake wear particle emissions

Similarly to tyre wear, brake wear emissions can be calculated by:

$$E_{BRAKE,i,j} = N_j \times M_j \times e_{BRAKE,\,TSP,j} \times f_{BRAKE,i} \times S(V)_{BRAKE} \qquad \text{(Equation B5)}$$

Where the nomenclature is similar to that of Equation B2.

TSP emission factors for brake wear particles are given in Table B4, together with the range and a quality code for the emission factor.

Table B4: TSP emission factors from brake wear.

Vehicle class (j)	Emission factor (g/km) $e_{BRAKE,TSP}$
Two-wheel vehicles	0.0037
Cars	0.0075
LGV	0.0117
HDV	Equation B6

The HDV emission factor is calculated by adjusting the passenger car emission factor to fit heavy-duty vehicle experimental data:

$$(e_{BRAKE})_{HDV} = 3.13 \cdot LCF_{BRAKE} \cdot (e_{BRAKE})_{PC} \qquad \text{(Equation B6)}$$

In Equation B6, 3.13 is an empirical factor derived from experimental data and $\boldsymbol{LCF_B}$ is defined in a similar way to $\boldsymbol{LCF_{TYRE}}$ and can be determined again by linear regression on experimental data by the equation:

$$\boldsymbol{LCF_{BRAKE} = 1 + 0.79 \times LF} \qquad \text{(Equation B7)}$$

LF again has the value of 0 for an empty truck and 1 for a fully laden one. Equations B6 and B7 are also used for urban buses and coaches. The mass fraction of TSP in the different particle size classes is shown in Table B5.

Table B5: Size distribution of brake wear emitted particles.

Particle size class (i)	Mass fraction (f_{BRAKE}) of TSP
TSP	1.000
PM_{10}	0.980
$PM_{2.5}$	0.390
PM_1	0.100
$PM_{0.1}$	0.080

The speed correction factor for the case of brake wear is given in Figure B2. In this case, the speed correction is normalised for a speed of 65 km/h, and the slope is generally larger than for tyre wear because brake wear is negligible at high motorway speeds when limited braking occurs. Again, although the proposed equation has been obtained from measurements on passenger cars, it is to be used for all vehicle categories.

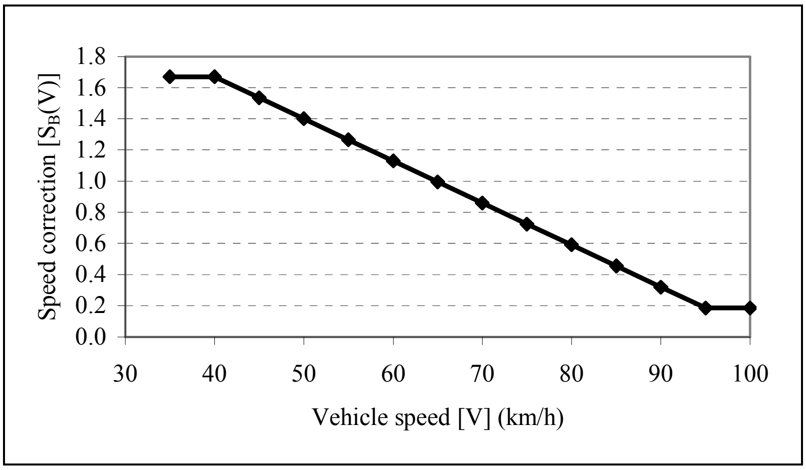

Figure E2: Speed correction factor ($\mathbf{S_{B(V)}}$) for brake wear particle emissions.

Road surface wear emissions

There is very little information on airborne emission rates from asphalt wear, and therefore the quality of the detailed methodology does not differ from the quality of the simple one. The detailed methodology only provides a mass-weighted size classification of road surface wear particles based on the work of Lükewille *et al.* (2001), according to the equation:

$$E_{ROAD,i,j} = N_j \times M_j \times (e_{ROAD})_j \times f_{ROAD,i} \qquad \text{(Equation B8)}$$

Where the nomenclature is similar to Equation B2. The TSP emission factors for particles of road surface wear are presented in Table B1. The mass fraction of TSP in the different particle size classes is shown in Table B6.

Table B6: Size distribution of road surface wear emitted particles.

Particle size class (i)	Mass fraction (f_{ROAD}) of TSP
TSP	1.00
PM_{10}	0.50
$PM_{2.5}$	0.27

Due to the lack of appropriate experimental data, no emission factors are included for road surface wear associated with the use of studded tyres, although it is recognised that in some countries this may be an important particle source. Preliminary values for road surface wear TSP emissions are shown in Table B7. These TSP values should correspond to primary particles from road surface wear but they are based on limited information and are highly uncertain.

Table B7: TSP emission factors from road surface wear.

Vehicle class (j)	Emission factor (g/km) $e_{ROAD,TSP}$
Two-wheel vehicles	0.0060
Cars	0.0150
LGV	0.0150
HDV	0.0760

Appendix C. HDM-4 tyre consumption model

The following text is adapted from Volume 7 of HDM-4 (Bennett and Greenwood, 2001), and from Carpenter and Cenek (1999).

C.1 HDM-4 Mechanistic Tyre Model

Model formulation

The HDM tyre consumption model is based upon the principles of slip energy. Tyre slip is the circumferential motion of the tyre relative to the wheel rim. Slip energy is the product of the total distance slipped by a tyre multiplied by the horizontal force on the tyre. HDM-4 uses the model formulation presented below from Carpenter and Cenek (1999):

$$\mathrm{TWT} = \mathrm{FLV}\left(\mathrm{C0tc} + \frac{\mathrm{Ctcte\ FNC\ CFT}^2}{\mathrm{NFT}} + \frac{\mathrm{Ctcte\ FNL\ LFT}^2}{\mathrm{NFT}}\right)$$

(Equation C1)

where

TWT = volumetric tread wear rate of a tyre (dm3/1000 tyre-km)
FLV = a factor for local effects, vehicle type, *etc.*
C0tc = a model coefficient
Ctcte = a model coefficient
FNC = a factor representing the variation in circumferential forces
FNL = a factor representing the variation in lateral forces
CFT = circumferential force on tyre (N)
LFT = lateral force on tyre (N)
NFT = the load on the tyre normal to the tyre-road contact area (N)

Since both acceleration and deceleration forces contribute to tyre wear, the terms CFT and LFT in Equation C1 represent the mean of the absolute values of CFT and LFT. The factors FNC and FNL were introduced by Carpenter and Cenek (1999) to account for the effects of random fluctuations, or noise, in the values of CFT and LFT. These changes are important since they allow the model to be used to predict the effects of traffic interactions on tyre consumption.

Quantification of model parameters

FLV

The factor FLV reflects localised effects on tyre consumption which would not be embodied in the standard model parameters, such as vehicle type, road roughness, macrotexture, aggregate abrasion, tyre type, weather, regional effects, *etc.* It effectively serves as a 'rotation' calibration factor. Carpenter and Cenek (1999) do not provide any details on values for FLV as a function of operating conditions so the value is set to 1.0 until further work is done in this area.

C0tc and Ctctc

On the basis of the previous work and their own experiments Carpenter and Cenek (1999) proposed the default tyre consumption model parameter values for the HDM-4 representative vehicles in Table C1.

Table C1: Carpenter and Cenek (1999) tyre consumption model parameters.

Representative Vehicle	Description	C0tc (dm^3/1000 km)	Ctcte (dm^3/MNm)
1	Motorcycle	0.001	0.0009
2	Small Car	0.001	0.0005
3	Medium Car	0.001	0.0005
4	Large Car	0.001	0.0005
5	Light Delivery Vehicle	0.001	0.0005
6	Light Goods Vehicle	0.001	0.0005
7	Four Wheel Drive	0.001	0.0005
8	Light Truck	0.001	0.0003
9	Medium Truck	0.001	0.0003
10	Heavy Truck	0.001	0.0003
11	Articulated Truck	0.001	0.0003
12	Mini Bus	0.001	0.0003
13	Light Bus	0.001	0.0003
14	Medium Bus	0.001	0.0003
15	Heavy Bus	0.001	0.0003
16	Coach	0.001	0.0003

However, when testing the model, the values for C0tc were found to be too low and resulted in unreasonably high tyre lives. Alternative values were quantified using estimated tyre lives. The mechanistic forces were calculated using the default HDM-4 model parameters for each of the representative vehicles. While reasonable parameters were obtained for some vehicles, it was not possible to use them with absolute confidence. The values of Ctcte used in this study for LDVs, HDVs and motorcycles were 0.0005, 0.0003 and 0.0009 dm^3/MNm respectively.

FNC

The factor FNC is a function of speed variation (Table C2). If a vehicle travels at a perfectly steady speed the value is 1.0. As the frequency and magnitude of the speed variations increase, the value of FNC increases. As shown in Carpenter and Cenek (1999), speed changes have the greatest effects at low speeds due to the inertial effects and effective mass. The effects are also proportional to mass, with heavy vehicles having the greatest impacts. For motorcycles the value is approximately half that of a passenger car, which in turn is approximately half that of a heavy truck. However we should note that the effect of vehicle type is substantially less than the effect of the level of speed variation. Also, one would not expect the heavy truck to be able to achieve a high level of speed variation in practice, and therefore it would not experience the highest FNC values listed above. FNC values are presented in Table C3 for three vehicle types: motorcycle, medium car, and heavy truck.

Table C2: Acceleration noise and speed range associated with three levels of speed variation.

	Acceleration Noise	Speed Variation Range
Low variation	0.1 m/s^2	-5 to +8 km/h
Medium variation	0.3 m/s^2	-12 to +24 km/h
High variation	0.6 m/s^2	-33 to +82 km/h

Table C3: Values of FNC for motorcycles, medium cars and heavy trucks.

Motorcycle

Mean Speed	**30 km/h**	**50 km/h**	**70 km/h**	**100 km/h**
Speed variation				
Low	1.1	1.1	1.05	1.05
Medium	1.9	1.7	1.5	1.4
High	7.2	4.6	3.2	2.2

Medium Car

Mean Speed	**30 km/h**	**50 km/h**	**70 km/h**	**100 km/h**
Speed variation				
Low	1.3	1.2	1.1	1.1
Medium	3.8	2.6	2.0	1.5
High	12.2	7.4	4.7	2.9

Heavy Truck

Mean Speed	**30 km/h**	**50 km/h**	**70 km/h**	**100 km/h**
Speed variation				
Low	1.7	1.4	1.2	1.1
Medium	7.6	4.9	3.2	2.1
High	29.4	16.6	9.8	4.9

a. Linear interpolation should be used for intermediate values of mean speed and speed variation.
b. For all cars, delivery vehicles and 4 wheel drives (vehicle numbers 2-7): FNC = FNCbasic
c. For motorcycles (vehicle number 1): FNC = 0.5 FNCbasic + 0.5
d. For all trucks and buses (vehicle numbers 8-16): FNC = 2 FNCbasic - 1

FNL

To estimate the value for **FNL**, Carpenter and Cenek (1999) used data from two routes with different severities in terms of the horizontal curvature (42 vs 425 degrees/km). Using appropriate values for FNC and TWT it was found that there was little variation in FNL between the routes, in spite of the large difference in rout severity. It was concluded that “FNL is likely to be relatively constant for many routes” and that “it is not likely to be significantly influenced by vehicle type”. On the basis of their analysis it was proposed that a constant value of 2.5 be adopted for all routes and all vehicle types.

CFT

This is the average absolute value of the circumferential force on the tyre. It is approximately the total tractive force for each tyre which is calculated in HDM4 (total tractive force/number of wheels, as in this analysis we are not distinguishing between the forces on the driven and undriven wheels). The following default values are proposed:

Medium-size car	190 N
Motorcycle	230 N
Heavy truck	500 N

LFT

This is the average lateral force on the tyre. An approximate value can be calculated if the average curvature of the road and average cornering speed are known. Typically the average cornering speed is about 0.8 of the average route speed.

Average road curvature = 100°/km
Average road radius, r = (360/100).1000/2π, r = 573m
Average cornering speed, Vc = 90x0.8/3.6 = 20m/s
LFT = m.Vc^2/r, where m = vehicle mass per wheel

Medium-size car	Vehicle mass per wheel, m = 300kg	LFT	=	209 N
Motorcycle	Vehicle mass per wheel, m = 100kg	LFT	=	70 N
Heavy truck	Vehicle mass per wheel, m = 1300kg	LFT	=	907 N

The average cornering speed is actually a function of the approach speed and the road radius. For this study, a function developed by Brodin and Carlsson (1986) was used to predict cornering speed (for all vehicle types), based on these input parameters. The default road radius of 573 m was used.

$$v_{mc} = \frac{1}{\sqrt{\left(\frac{1}{va}\right)^2 + 0.15\left(\frac{1}{R} - 0.001\right)}}$$

where v_{mc} is the median curve speed in m/s
v_a is the median approach speed in m/s

(Equation C2)

NFT

The vehicle weight on the tyre = ***mxg***, where ***g*** = acceleration due to gravity = 9.81 m/s^2

Default values:

Medium-size car	2943 N
Motorcycle	981 N
Heavy truck	12753 N

Further work is required to establish model parameters which will give sensible predictions with the mechanistic tyre model. Until this is done, the interim solution described in the following section was adopted for HDM-4.

C.2 Interim HDM-4 Tyre Model

Due to the problems with the proposed new mechanistic tyre model, an interim model was adopted for HDM-4 v 1.0 based on the HDM-III model (Watanatada, *et al.*, 1987).

The rate of tread wear is calculated as:

TWT = C0tc + Ctcte TE (Equation C3)

where :

C0tc is a model coefficient for tread wear in $dm^3/1000$ km
Ctcte is a model coefficient
TE is the tangential energy in Jm

The tangential energy, circumferential, lateral and normal forces on the tyres are calculated as:

$$TE = \frac{CFT^2 + LFT^2}{NFT} \quad C4$$

$$CFT = \frac{(1 + CTCON\ dFUEL)(Fa + Fr + Fg)}{NUM\_WHEELS} \quad C5$$

$$LFT = \frac{Fc}{NUM\_WHEELS} \quad C6$$

$$NFT = \frac{M\ g}{NUM\_WHEELS} \quad C7$$

Where:

Fa	is the aerodynamic force opposing motion in N
Fr	is the rolling resistance in N
Fg	is the gradient force in N
dFUEL	is the additional fuel due to congestion as a fraction
NUM_WHEELS	is the number of wheels on the vehicle
Fc	is the curvature resistance in N
M	is the vehicle mass
g	is the acceleration due to gravity in m/s^2

Values for use with this tyre model for different vehicle categories are provided in HDM-4. However, this involves the separate calculation of many different parameters, and many of the routines involved cannot be easily extracted.

In any case, although this is an interim solution, Bennett and Greenwood (2001) state there are a number of unsatisfactory issues with the above model. Accordingly, the model's predictions should be viewed with caution until a more rigorous model is implemented.

Appendix D. USEPA AP-42 model

Re-entrained road dust emission factors are calculated for both unpaved and paved roads include all PM sources (exhaust, tyre wear, brake wear, road surface wear and resuspension). The models did not formerly apply to days with rain, although in a memorandum the USEPA (2003) indicated that a 25% reduction in emissions should be assumed for such days.

For unpaved roads, the following equation is used:

$$EF_{UNPAVED} = F_{UNPAVED} \times 5.9 \times \frac{sL}{12} \times \frac{V}{30} \times \left[\frac{W}{3}\right]^{0.7} \times \left[\frac{Nw}{4}\right]^{0.5} \times \frac{(365-d)}{365} \times 453.592 \qquad \text{(Equation D1)}$$

Where:

$EF_{UNPAVED}$	=	Fleet average unpaved road dust emission factor (g/mile)
$F_{UNPAVED}$	=	The fraction of particles less than or equal to the particle size cut-off (Table D3)
sL	=	Silt content (particles <75 μm diameter) of the surface material (%)
V	=	Fleet average vehicle speed (mph)
W	=	Fleet average vehicle weight (imperial tons)
N_w	=	Fleet average number of wheels per vehicle
d	=	Average number of days per year with more than 0.01 inches of rain

Table D3: Fraction of particles for each cut-off.

Particle size cut-off (μm)	$F_{UNPAVED}$
10	0.36
5	0.20
2.5	0.095

For paved roads, the following equation is used:

$$EF_{PAVED} = e_{PAVED} \times \left[\frac{sL}{2}\right]^{0.65} \times \left[\frac{W}{3}\right]^{1.5} \qquad \text{(Equation D2)}$$

Where:

EF_{PAVED}	=	Fleet average paved road dust emission factor (g/mile)
e_{PAVED}	=	The base emission factor for the particle size cut-off
sL	=	The road surface silt loading (g/m^2)
W	=	Fleet average vehicle weight (tons)

For size cut-off of 10 μm, e_{PAVED} is 7.3 g/mile, and for a cut off of 2.5 μm e_{PAVED} is 3.3 g/mile.

There is little information on silt loadings of UK roads, but Ball and Caswell (1983) give data that would equate to about 0.02 g/m^2 as an upper limit (cited in QUARG, 1996).

Appendix E. German traffic situation model

An alternative version of the USEPA model for paved roads was developed by Rauterberg-Wulff (2000) for use in Berlin. The further modification to the AP-42 method proposed by Gamez *et al.* (2001), and further modified by Düring *et al.* (2002), involved the separation of the exhaust and non-exhaust contributions. By 2004, the modified AP-42 method had been abandoned in favour of allocating emission factors to 'traffic situations' (Lohmeyer *et al.*, 2004a). Examples of these emission factors are given in the Table E1. The traffic situations are taken from the Handbook of emission factors (INFRAS, 2004). The emission factors provided are for flat terrain and typical rainfall in
Germany.

Table E1: Total non-exhaust PM_{10} emission factors by traffic situation (Lohmeyer et al., 2004a).

Traffic situation (INFRAS, 2004)	Location	Description	Speed limit (km/h)	PM_{10} emission factor (g/vkm)	
				Cars and LGVs	HDVs
AB>120	Motorway	Motorway, no speed limit	-	22	200
AB_120	Motorway	Motorway,120 km/h speed limit	120	22	200
AB_100	Motorway	Motorway, 100 km/h speed limit	100	22	200
AB_80	Motorway	Motorway, 80 km/h speed limit	80	22	200
AB_60	Motorway	Motorway, 60 km/h speed limit	60	22	200
AB_StGo	Motorway	Motorway stop-and-go	-	22	200
Tunnel AB_100	Motorway	Tunnel, motorway, 100 km/h speed limit	100	10	200
Tunnel AB_80	Motorway	Tunnel, motorway, 80 km/h speed limit	80	10	200
Tunnel AB_60	Motorway	Tunnel, motorway, 60 km/h speed limit	60	10	200
AO1	Rural	Well-developed, straight	100	22	200
AO2	Rural	Well-developed, even bends	100	22	200
AO3	Rural	Uneven bends	100	22	200
Tunnnel, IO_HVS>50	Urban	Tunnel, city, speed limit > 50 km/h	60	10	200
IO_HVS>50	Urban	City, speed limit >50 km/h	60	22	200
HVS1	Urban	Main through road, right of way, no hold	50	22	200
HVS2	Urban	Main road, right of way, minimal hold ups	50	30	300
HVS3	Urban	Main road, right of way, medium hold ups	50	40	380
HVS4	Urban	Main road, right of way, major hold ups	50	50	450
LAS1	Urban	Main road, traffic lights, minimum delay	50	40	380
LSA2	Urban	Main road, traffic lights, medium delay	50	60	600
LSA3	Urban	Main road, traffic lights, heavy delay	50	90	800
IO_Kem	Urban	City centre	50	90	800
IO_NS_dicht	Urban	Side-road, self-contained development	50	90	800

Appendix F. SMHI model

A model was developed by the Swedish Environmental and Health Protection Administration for resuspension, proposed by Johansson *et al*. (1998) (cited in Rauterberg-Wulff, 2000 and Gustafsson, 2003). This model was briefly described by Boulter (2005a), but has since been updated (Omstedt *et al.*, 2005). The new version of the model is described in outline below; the full version is considerably more complex.

$$e_f^{tot} = e_f^{direct} + e_f^{resuspension} \qquad \text{(Equation F1)}$$

Where: e_f^{tot} = Total PM emission factor
e_f^{direct} = Emission factor for exhaust particles
$e_f^{resuspension}$ = Emission factor for resuspended particles

For the $e_f^{resuspension}$ term, different equations are presented for winter and summer:

$$\textit{Winter:} \quad e_f^{resuspension} = f_q \,.\, d \,.\, e_f^{ref,winter} \qquad \text{(Equation F2)}$$

$$\textit{Winter:} \quad e_f^{resuspension} = f_q \,.\, e_f^{ref,summer} \qquad \text{(Equation F3)}$$

Where: f_q = Source strength for resuspension, which is related to the moisture content of the road dust.
d = Amount of dust on the road.
e_f^{ref} = Reference emission factor.

A critical parameter in this model is the reference emission factor. It sets a baseline for the model and should be estimated for situations with high suspension. Emission factors for particles can be estimated using tracer methods if background and roadside concentrations of particles and NO_x are known:

$$e_f^{PM} = e_f^{NO_x} \left(\frac{C_{PM}^{roadside} - C_{PM}^{background}}{C_{NO_x}^{roadside} - C_{PM}^{background}} \right)$$

Equation F4

Appendix G. VLUFT model

There is little documentation in English describing the VLUFT model. The following form of the model was presented by Tønnesen (2003) for dry roads.

$$Q_{PM10} = Q_{EP} + \left[[Q_{R2.5} + (c \times ((a \times TT) + b)) \times \left(\frac{V_D}{V_{D,ref}} \right)^2] \times [(0.98 \times S_T) + 0.02] \right]$$

(Equation G1)

Where:

Q_{PM10}	=	PM_{10} emission factor from all sources (g/vkm).
Q_{EP}	=	Average exhaust particle emission factor (g/vkm). Assumed to be $PM_{2.5}$, and to be calculated using an emission model.
$Q_{R2.5}$	=	Emission of all non-exhaust PM2.5 (g/vkm). Assumed to be 0.4 x ***c***.
c	=	Empirical factor to determine the coarse fraction PM_{10}-$PM_{2.5}$ (for Oslo, ***c*** = 0.24 g/vkm).
a and b	=	Derived linear constants for heavy traffic (for Oslo, ***a*** = 0.258, ***b*** = 1.436).
TT	=	Percentage of heavy vehicles (% of vehicles with weight > 3.5 tons).
V_D	=	Driving speed (km/h).
$V_{D,ref}$	=	Reference driving speed (70 km/h).
S_T	=	Fraction of in-use studded tyres.

In order to quantify the source strength of the resuspended roadside dust, some basic assumptions were made. The dependency of emission strength on the percentage of heavy vehicles for resuspended road dust was assumed to be linear, and the dependency on the average driving speed was assumed to be quadratic. In addition, the amount of dust available for resuspension was assumed to be linearly dependent on the use of studded tyres, decreasing from 1 to 0.02 with decreasing studded tyre use from 100% to 0%. For a roadside measurement site it was assumed that the emission ratio for coarse fraction dust (PM_{10} – $PM_{2.5}$) to fine fraction dust ($PM_{2.5}$) would be directly proportional to the measured concentration ratio during hours with high concentration levels. Based on roadside measurements of particles, it was further assumed that near all of the coarse fraction dust would originate from the road surface, and that the amount of fine fraction dust from road surface was small compared with the coarse fraction.

Hourly measurements of roadside concentrations of PM_{10}, $PM_{2.5}$, traffic volume, traffic speed and heavy vehicle fraction made in Oslo were used to determine the linear coefficients for the dependency of the heavy vehicle fraction. This was achieved by a comparison of the concentration ratios for coarse fraction dust versus fine fraction dust for different heavy vehicle fractions.

Appendix H. Model from DAPPLE project

Patra *et al*. (2005) found that the flux of particulate material to air is approximately 14% (with a range of 5-35%) of the sum of the fluxes along and across the road.

The formula summarising the results is:

$$\begin{aligned} \boldsymbol{F_z} &= \boldsymbol{k\,(F_x + F_y)} \\ &= \boldsymbol{k\,(F_x + k_1 F_x)} \\ &= \boldsymbol{k\,F_x(1 + k_1)} \end{aligned}$$

Where

$\boldsymbol{F_x}$ = amount of material removed from a road segment along the road per unit time by the whole traffic (g/s)

$\boldsymbol{F_y}$ = amount of material removed from a road segment across the road per unit time by the whole traffic (g/s)

$\boldsymbol{F_z}$ = amount of material removed from a road segment to air per unit time by the whole traffic (g/s)

$$\boldsymbol{k = \frac{f'_r}{f'_x + f'_y}}$$

$$\boldsymbol{k_1 = \frac{f'_y}{f'_x}}$$

Where

$\boldsymbol{f'_x}$ = amount of material removed from a road segment along the road per unit time by one vehicle (g/s)

$\boldsymbol{f'_y}$ = amount of material removed from a road segment to across the road per unit time by one vehicle (g/s)

For particles in the size range 0.75-10 µm, average values for these parameters are given in Table H1.

Table H1: Flux estimates (Patra *et al.,* 2005).

Parameter	Estimated value	
	Average	Range
$\boldsymbol{F_x}$	16 g/s	-
$\boldsymbol{F_y}$	2.0 g/s	-
$\boldsymbol{F_z}$	2.5 g/s	0.64 – 15 g/s
$\boldsymbol{f'_x}$	12 g/veh	8.1 – 22 g/veh
$\boldsymbol{f'_y}$	1.6 g/veh	1.0 – 2.8 g/veh

Patra *et al.* (2005) recommend that the average rate of wear of London's road surface should be estimated from highways engineering data, and that emission of 35% of this to roadside air be considered as a first estimate of additional paved road surface emissions in the London Atmospheric Emissions Inventory.

Appendix I. Emission estimates for the UK – EMEP method

Total UK emissions of PM_{10}, $PM_{2.5}$, PM_1 and PM_{coarse} on urban roads, rural roads and motorways, as calculated using the EMP method, are shown for five reference years in Tables I1 to I11. As no emission factors are given in EMEP for PM1 emissions due to road surface wear, no results have been reported. For PM_{10} emissions the full time series between 1970 and 2005, plus 2010, 2015 and 2025, are shown in Figures I1 to I9.

Table I1: UK PM_{10} emissions due to tyre wear in five reference years.

Vehicle category	Road type	PM_{10} emissions by year (tonnes/year)				
		2000	2005	2010	2015	2025
Cars	Urban roads	1402.5	1496.2	1689.2	1711.0	1793.7
	Rural Roads	1106.4	1164.4	1394.7	1444.5	1549.4
	Motorways	391.3	442.7	499.6	517.5	555.0
LGVs	Urban roads	287.0	334.6	422.1	467.0	525.7
	Rural Roads	248.7	311.3	391.1	433.0	488.5
	Motorways	88.2	110.0	127.1	140.5	158.6
Rigid HGVs	Urban roads	88.4	89.7	84.1	85.8	89.3
	Rural Roads	130.4	133.3	129.7	134.1	139.9
	Motorways	59.8	64.7	59.4	61.6	64.2
Articulated HGVs	Urban roads	50.1	54.8	61.7	66.5	71.9
	Rural Roads	210.7	236.9	264.0	285.9	310.1
	Motorways	218.3	255.8	272.5	295.4	320.2
Buses	Urban roads	62.5	53.4	49.5	49.5	49.5
	Rural Roads	28.1	29.1	26.9	26.9	26.9
	Motorways	8.3	8.7	8.1	8.1	8.1
Motorcycles	Urban roads	8.4	10.8	12.0	12.4	12.4
	Rural Roads	5.7	7.0	7.7	8.0	8.0
	Motorways	1.2	1.2	1.4	1.4	1.4
Total		4395.9	4804.6	5500.9	5749.0	6172.8

Table I2: UK PM_{10} emissions due to brake wear in five reference years.

Vehicle category	Road type	PM_{10} emissions by year (tonnes/year)				
		2000	2005	2010	2015	2025
Cars	Urban roads	1874.5	1999.6	2257.6	2286.8	2397.3
	Rural Roads	899.2	946.4	1133.6	1174.0	1259.3
	Motorways	91.9	104.0	117.3	121.5	130.3
LGVs	Urban roads	378.8	441.7	557.2	616.4	694.0
	Rural Roads	199.7	249.9	314.0	347.6	392.1
	Motorways	20.5	25.5	29.5	32.6	36.8
Rigid HGVs	Urban roads	215.5	220.6	206.9	211.0	219.5
	Rural Roads	203.0	205.8	200.3	207.1	215.9
	Motorways	36.8	39.5	36.3	37.6	39.2
Articulated HGVs	Urban roads	55.9	59.4	66.9	72.1	77.9
	Rural Roads	150.2	166.3	185.3	200.7	217.7
	Motorways	60.0	69.3	73.9	80.1	86.8
Buses	Urban roads	152.4	131.3	121.6	121.6	121.6
	Rural Roads	43.7	44.9	41.5	41.5	41.5
	Motorways	5.1	5.3	5.0	5.0	5.0
Motorcycles	Urban roads	12.8	16.6	18.4	19.1	19.0
	Rural Roads	5.3	6.5	7.2	7.5	7.5
	Motorways	0.3	0.3	0.4	0.4	0.4
Total		4405.7	4732.9	5372.8	5582.3	5961.8

Table I3: UK PM_{10} emissions due to road surface wear in five reference years.

Vehicle category	Road type	PM_{10} emissions by year (tonnes/year)				
		2000	2005	2010	2015	2025
Cars	Urban roads	1203.7	1284.1	1449.8	1468.5	1539.4
	Rural Roads	1220.2	1284.2	1538.2	1593.1	1708.8
	Motorways	506.7	573.4	647.1	670.2	718.8
LGVs	Urban roads	155.9	181.8	229.4	253.7	285.7
	Rural Roads	173.7	217.3	273.1	302.3	341.1
	Motorways	72.3	90.2	104.2	115.2	130.1
Rigid HGVs	Urban roads	160.6	164.4	154.2	157.2	163.6
	Rural Roads	288.6	292.6	284.7	294.3	307.0
	Motorways	164.0	175.7	161.5	167.3	174.3
Articulated HGVs	Urban roads	41.7	44.3	49.8	53.7	58.1
	Rural Roads	213.6	236.4	263.4	285.3	309.5
	Motorways	266.9	308.7	328.8	356.4	386.3
Buses	Urban roads	113.6	97.8	90.6	90.6	90.6
	Rural Roads	62.1	63.8	59.0	59.0	59.0
	Motorways	22.9	23.7	22.1	22.1	22.1
Motorcycles	Urban roads	6.7	8.6	9.6	9.9	9.9
	Rural Roads	5.9	7.2	7.9	8.2	8.2
	Motorways	1.4	1.5	1.6	1.7	1.7
Total		4680.4	5055.5	5675.0	5908.8	6314.1

Table I4: UK $PM_{2.5}$ emissions due to tyre wear in five reference years.

Vehicle category	Road type	PM_{10} emissions by year (tonnes/year)				
		2000	2005	2010	2015	2025
Cars	Urban roads	981.8	1047.3	1182.5	1197.7	1255.6
	Rural Roads	774.5	815.0	976.3	1011.1	1084.6
	Motorways	273.9	309.9	349.7	362.2	388.5
LGVs	Urban roads	200.9	234.2	295.5	326.9	368.0
	Rural Roads	174.1	217.9	273.8	303.1	341.9
	Motorways	61.7	77.0	88.9	98.4	111.0
Rigid HGVs	Urban roads	61.9	62.8	58.9	60.1	62.5
	Rural Roads	91.3	93.3	90.8	93.9	97.9
	Motorways	41.9	45.3	41.6	43.1	44.9
Articulated HGVs	Urban roads	35.1	38.4	43.2	46.6	50.3
	Rural Roads	147.5	165.8	184.8	200.1	217.1
	Motorways	152.8	179.1	190.8	206.8	224.1
Buses	Urban roads	43.8	37.4	34.6	34.6	34.6
	Rural Roads	19.6	20.3	18.8	18.8	18.8
	Motorways	5.8	6.1	5.7	5.7	5.7
Motorcycles	Urban roads	5.9	7.6	8.4	8.7	8.7
	Rural Roads	4.0	4.9	5.4	5.6	5.6
	Motorways	0.8	0.9	1.0	1.0	1.0
Total		3077.2	3363.2	3850.6	4024.3	4320.9

Table I5: UK $PM_{2.5}$ emissions due to brake wear in five reference years.

Vehicle category	Road type	PM_{10} emissions by year (tonnes/year)				
		2000	2005	2010	2015	2025
Cars	Urban roads	746.0	795.8	898.4	910.0	954.0
	Rural Roads	357.9	376.6	451.1	467.2	501.2
	Motorways	36.6	41.4	46.7	48.4	51.9
LGVs	Urban roads	150.8	175.8	221.7	245.3	276.2
	Rural Roads	79.5	99.4	125.0	138.3	156.0
	Motorways	8.1	10.2	11.7	13.0	14.6
Rigid HGVs	Urban roads	85.8	87.8	82.3	84.0	87.4
	Rural Roads	80.8	81.9	79.7	82.4	85.9
	Motorways	14.7	15.7	14.4	15.0	15.6
Articulated HGVs	Urban roads	22.2	23.6	26.6	28.7	31.0
	Rural Roads	59.8	66.2	73.7	79.9	86.6
	Motorways	23.9	27.6	29.4	31.9	34.5
Buses	Urban roads	60.6	52.2	48.4	48.4	48.4
	Rural Roads	17.4	17.9	16.5	16.5	16.5
	Motorways	2.0	2.1	2.0	2.0	2.0
Motorcycles	Urban roads	5.1	6.6	7.3	7.6	7.6
	Rural Roads	2.1	2.6	2.9	3.0	3.0
	Motorways	0.1	0.1	0.1	0.2	0.2
Total		1753.3	1883.5	2138.2	2221.5	2372.6

Table I6: UK $PM_{2.5}$ emissions due to road surface wear in five reference years.

Vehicle category	Road type	PM_{10} emissions by year (tonnes/year)				
		2000	2005	2010	2015	2025
Cars	Urban roads	650.0	693.4	782.9	793.0	831.3
	Rural Roads	658.9	693.4	830.6	860.3	922.7
	Motorways	273.6	309.6	349.4	361.9	388.2
LGVs	Urban roads	84.2	98.2	123.9	137.0	154.3
	Rural Roads	93.8	117.4	147.5	163.3	184.2
	Motorways	39.1	48.7	56.3	62.2	70.2
Rigid HGVs	Urban roads	86.7	88.8	83.3	84.9	88.3
	Rural Roads	155.9	158.0	153.7	158.9	165.8
	Motorways	88.5	94.9	87.2	90.3	94.1
Articulated HGVs	Urban roads	22.5	23.9	26.9	29.0	31.3
	Rural Roads	115.3	127.7	142.2	154.0	167.1
	Motorways	144.1	166.7	177.5	192.4	208.6
Buses	Urban roads	61.3	52.8	48.9	48.9	48.9
	Rural Roads	33.5	34.4	31.9	31.9	31.9
	Motorways	12.4	12.8	11.9	11.9	11.9
Motorcycles	Urban roads	3.6	4.7	5.2	5.4	5.4
	Rural Roads	3.2	3.9	4.3	4.4	4.4
	Motorways	0.8	0.8	0.9	0.9	0.9
Total		2527.4	2730.0	3064.5	3190.7	3409.6

Table I7: UK PM_1 emissions due to tyre wear in five reference years.

Vehicle category	Road type	PM_{10} emissions by year (tonnes/year)				
		2000	2005	2010	2015	2025
Cars	Urban roads	140.3	149.6	168.9	171.1	179.4
	Rural Roads	110.6	116.4	139.5	144.4	154.9
	Motorways	39.1	44.3	50.0	51.7	55.5
LGVs	Urban roads	28.7	33.5	42.2	46.7	52.6
	Rural Roads	24.9	31.1	39.1	43.3	48.8
	Motorways	8.8	11.0	12.7	14.1	15.9
Rigid HGVs	Urban roads	8.8	9.0	8.4	8.6	8.9
	Rural Roads	13.0	13.3	13.0	13.4	14.0
	Motorways	6.0	6.5	5.9	6.2	6.4
Articulated HGVs	Urban roads	5.0	5.5	6.2	6.7	7.2
	Rural Roads	21.1	23.7	26.4	28.6	31.0
	Motorways	21.8	25.6	27.3	29.5	32.0
Buses	Urban roads	6.3	5.3	4.9	4.9	4.9
	Rural Roads	2.8	2.9	2.7	2.7	2.7
	Motorways	0.8	0.9	0.8	0.8	0.8
Motorcycles	Urban roads	0.8	1.1	1.2	1.2	1.2
	Rural Roads	0.6	0.7	0.8	0.8	0.8
	Motorways	0.1	0.1	0.1	0.1	0.1
Total		439.6	480.5	550.1	574.9	617.3

Table I8: UK PM_1 emissions due to brake wear in five reference years.

Vehicle category	Road type	PM_{10} emissions by year (tonnes/year)				
		2000	2005	2010	2015	2025
Cars	Urban roads	191.3	204.0	230.4	233.3	244.6
	Rural Roads	91.8	96.6	115.7	119.8	128.5
	Motorways	9.4	10.6	12.0	12.4	13.3
LGVs	Urban roads	38.7	45.1	56.9	62.9	70.8
	Rural Roads	20.4	25.5	32.0	35.5	40.0
	Motorways	2.1	2.6	3.0	3.3	3.8
Rigid HGVs	Urban roads	22.0	22.5	21.1	21.5	22.4
	Rural Roads	20.7	21.0	20.4	21.1	22.0
	Motorways	3.8	4.0	3.7	3.8	4.0
Articulated HGVs	Urban roads	5.7	6.1	6.8	7.4	7.9
	Rural Roads	15.3	17.0	18.9	20.5	22.2
	Motorways	6.1	7.1	7.5	8.2	8.9
Buses	Urban roads	15.6	13.4	12.4	12.4	12.4
	Rural Roads	4.5	4.6	4.2	4.2	4.2
	Motorways	0.5	0.5	0.5	0.5	0.5
Motorcycles	Urban roads	1.3	1.7	1.9	1.9	1.9
	Rural Roads	0.5	0.7	0.7	0.8	0.8
	Motorways	0.0	0.0	0.0	0.0	0.0
Total		449.6	482.9	548.2	569.6	608.3

Table I9: UK PM_{coarse} emissions due to tyre wear in five reference years.

Vehicle category	Road type	PM_{10} emissions by year (tonnes/year)				
		2000	2005	2010	2015	2025
Cars	Urban roads	420.8	448.9	506.8	513.3	538.1
	Rural Roads	331.9	349.3	418.4	433.3	464.8
	Motorways	117.4	132.8	149.9	155.2	166.5
LGVs	Urban roads	86.1	100.4	126.6	140.1	157.7
	Rural Roads	74.6	93.4	117.3	129.9	146.5
	Motorways	26.5	33.0	38.1	42.2	47.6
Rigid HGVs	Urban roads	26.5	26.9	25.2	25.7	26.8
	Rural Roads	39.1	40.0	38.9	40.2	42.0
	Motorways	17.9	19.4	17.8	18.5	19.2
Articulated HGVs	Urban roads	15.0	16.5	18.5	20.0	21.6
	Rural Roads	63.2	71.1	79.2	85.8	93.0
	Motorways	65.5	76.8	81.8	88.6	96.1
Buses	Urban roads	18.8	16.0	14.8	14.8	14.8
	Rural Roads	8.4	8.7	8.1	8.1	8.1
	Motorways	2.5	2.6	2.4	2.4	2.4
Motorcycles	Urban roads	2.5	3.2	3.6	3.7	3.7
	Rural Roads	1.7	2.1	2.3	2.4	2.4
	Motorways	0.3	0.4	0.4	0.4	0.4
Total		1318.8	1441.4	1650.3	1724.7	1851.8

Table I10: UK PM_{coarse} emissions due to brake wear in five reference years.

Vehicle category	Road type	PM_{10} emissions by year (tonnes/year)				
		2000	2005	2010	2015	2025
Cars	Urban roads	1128.5	1203.8	1359.2	1376.7	1443.2
	Rural Roads	541.4	569.8	682.5	706.8	758.2
	Motorways	55.3	62.6	70.6	73.2	78.5
LGVs	Urban roads	228.1	265.9	335.4	371.1	417.8
	Rural Roads	120.2	150.4	189.0	209.3	236.1
	Motorways	12.3	15.4	17.7	19.6	22.1
Rigid HGVs	Urban roads	129.7	132.8	124.6	127.0	132.2
	Rural Roads	122.2	123.9	120.6	124.7	130.0
	Motorways	22.2	23.8	21.8	22.6	23.6
Articulated HGVs	Urban roads	33.7	35.8	40.3	43.4	46.9
	Rural Roads	90.4	100.1	111.6	120.8	131.1
	Motorways	36.1	41.7	44.5	48.2	52.2
Buses	Urban roads	91.7	79.0	73.2	73.2	73.2
	Rural Roads	26.3	27.0	25.0	25.0	25.0
	Motorways	3.1	3.2	3.0	3.0	3.0
Motorcycles	Urban roads	7.7	10.0	11.1	11.5	11.5
	Rural Roads	3.2	3.9	4.3	4.5	4.5
	Motorways	0.2	0.2	0.2	0.2	0.2
Total		2652.4	2849.4	3234.6	3360.8	3589.2

Table I11: UK PM_{coarse} emissions due to road surface wear in five reference years.

Vehicle category	Road type	PM_{10} emissions by year (tonnes/year)				
		2000	2005	2010	2015	2025
Cars	Urban roads	553.7	590.7	666.9	675.5	708.1
	Rural Roads	561.3	590.7	707.6	732.8	786.0
	Motorways	233.1	263.7	297.6	308.3	330.7
LGVs	Urban roads	71.7	83.6	105.5	116.7	131.4
	Rural Roads	79.9	100.0	125.6	139.1	156.9
	Motorways	33.3	41.5	47.9	53.0	59.8
Rigid HGVs	Urban roads	73.9	75.6	70.9	72.3	75.2
	Rural Roads	132.8	134.6	131.0	135.4	141.2
	Motorways	75.4	80.8	74.3	76.9	80.2
Articulated HGVs	Urban roads	19.2	20.4	22.9	24.7	26.7
	Rural Roads	98.2	108.7	121.2	131.2	142.4
	Motorways	122.8	142.0	151.2	163.9	177.7
Buses	Urban roads	52.2	45.0	41.7	41.7	41.7
	Rural Roads	28.6	29.3	27.1	27.1	27.1
	Motorways	10.5	10.9	10.2	10.2	10.2
Motorcycles	Urban roads	3.1	4.0	4.4	4.6	4.6
	Rural Roads	2.7	3.3	3.7	3.8	3.8
	Motorways	0.6	0.7	0.8	0.8	0.8
Total		2153.0	2325.5	2610.5	2718.0	2904.5

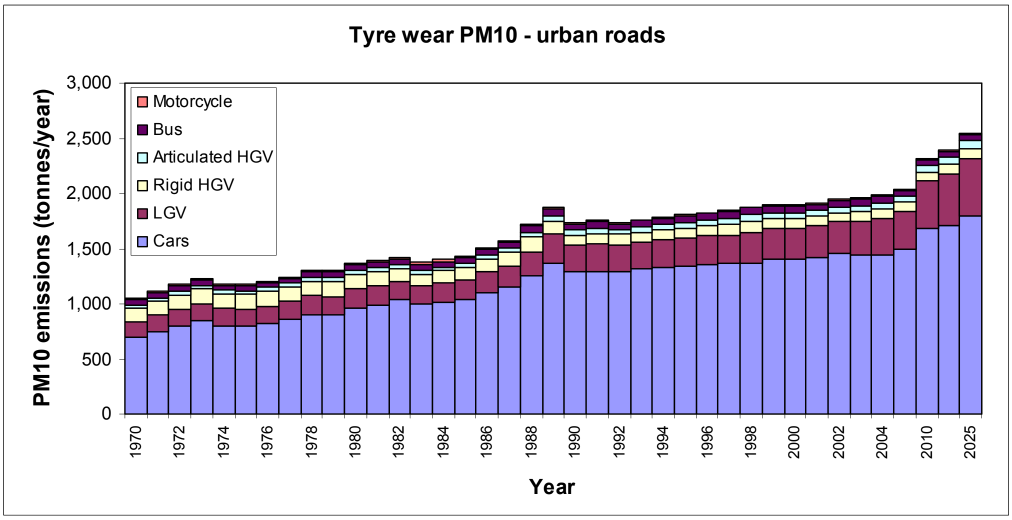

Figure I1: Total PM_{10} emissions due to tyre wear on urban roads in the UK (EMEP method).

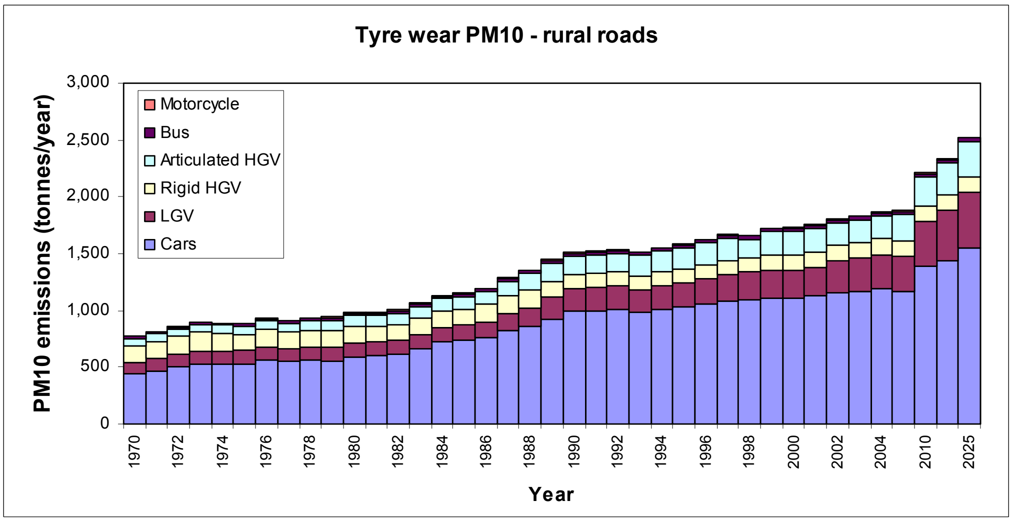

Figure I2: Total PM_{10} emissions due to tyre wear on rural roads in the UK (EMEP method).

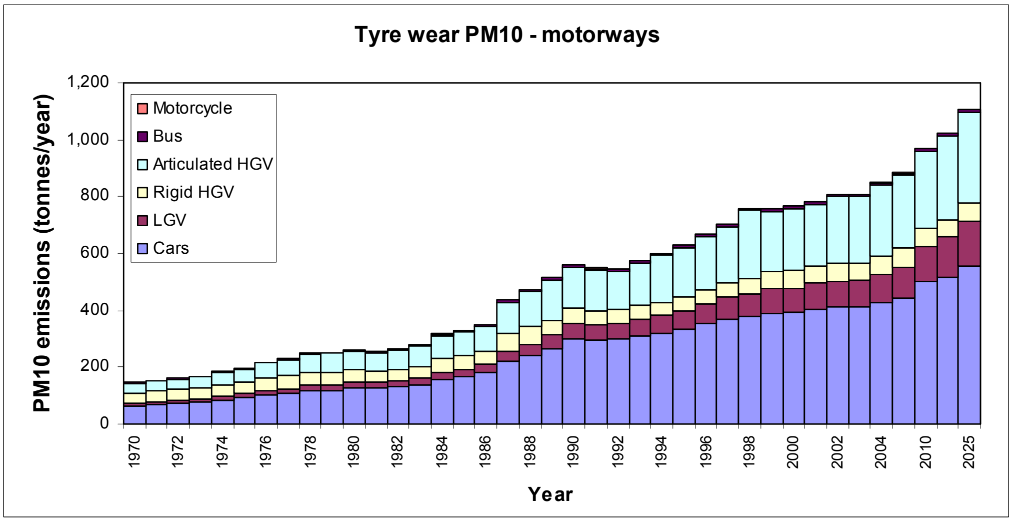

Figure I3: Total PM_{10} emissions due to tyre wear on motorways in the UK (EMEP method).

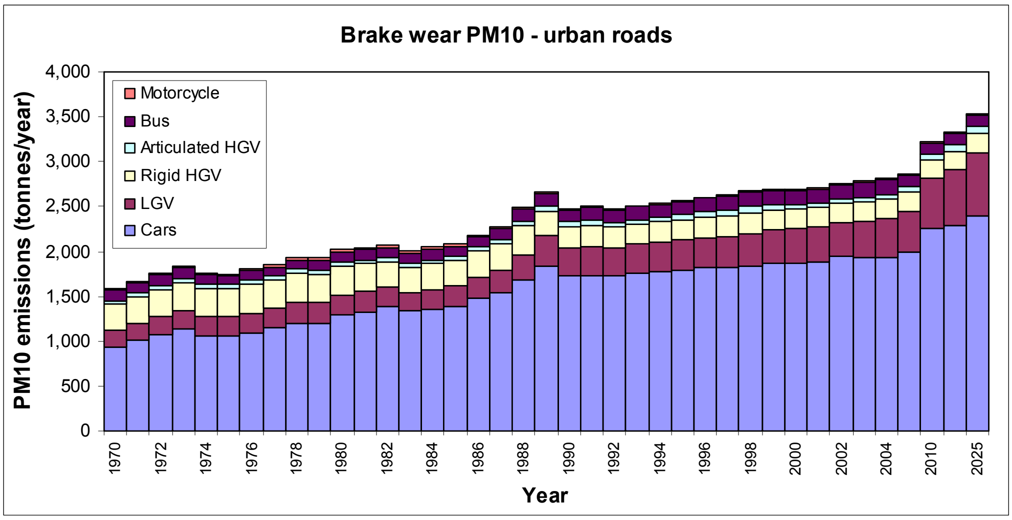

Figure I4: Total PM_{10} emissions due to brake wear on urban roads in the UK (EMEP method).

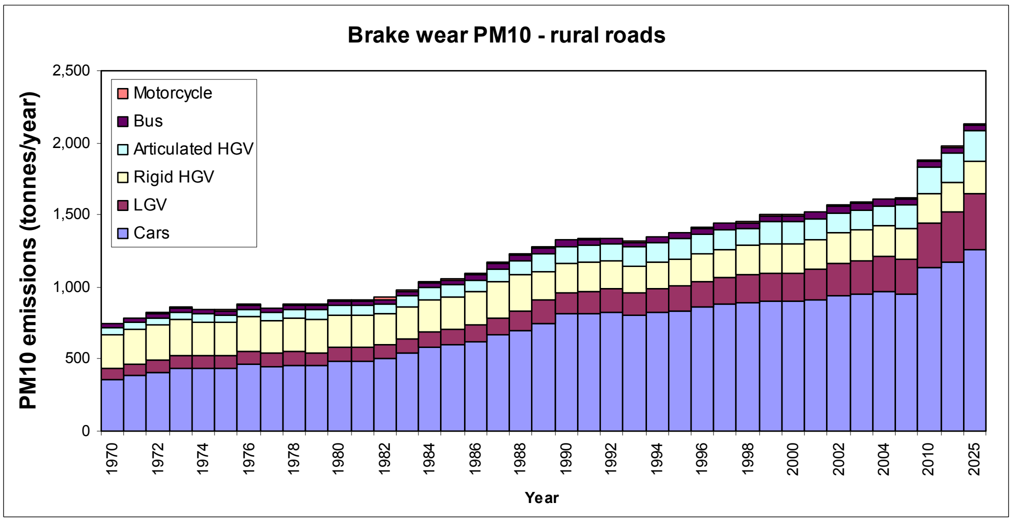

Figure I5: Total PM_{10} emissions due to brake wear on rural roads in the UK (EMEP method).

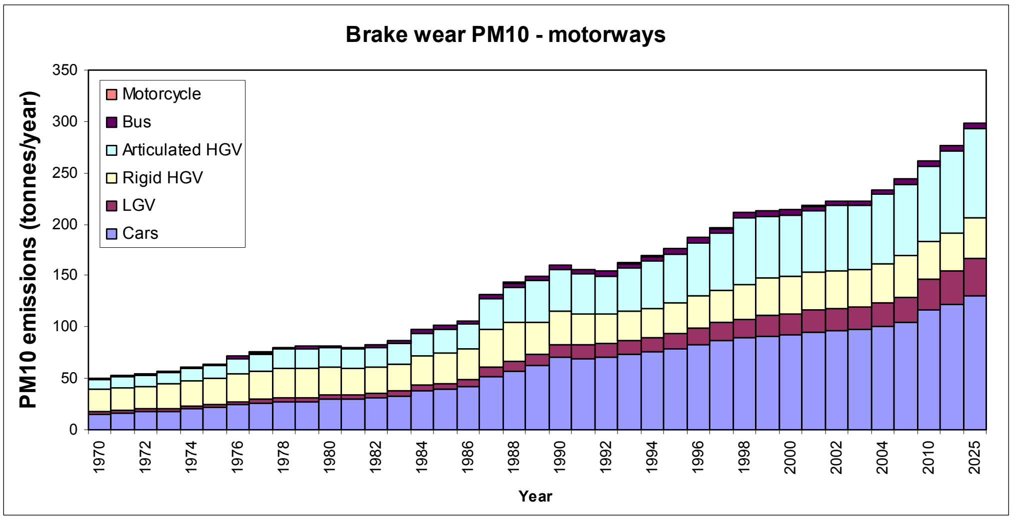

Figure I6: Total PM_{10} emissions due to brake wear on motorways in the UK (EMEP method).

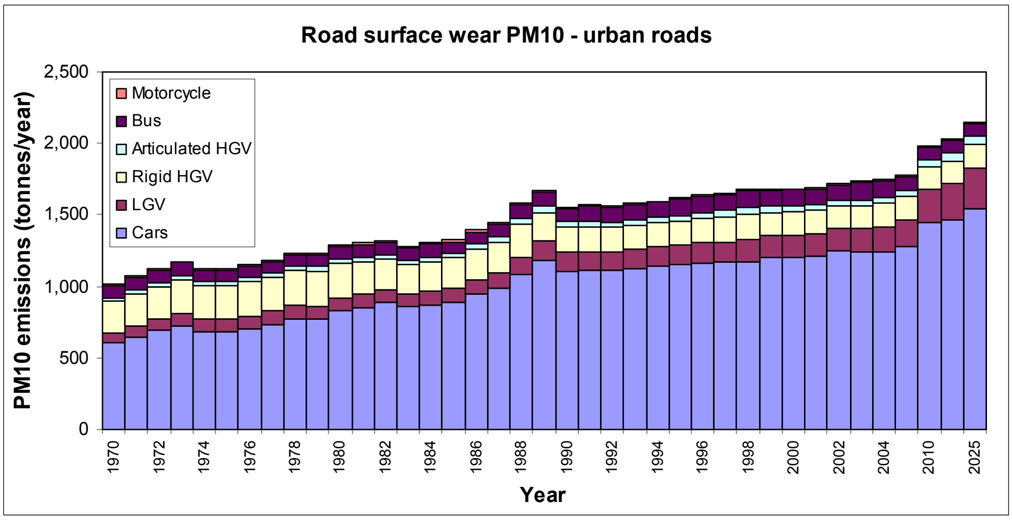

Figure I7: Total PM_{10} emissions due to road surface wear on urban roads in the UK (EMEP method).

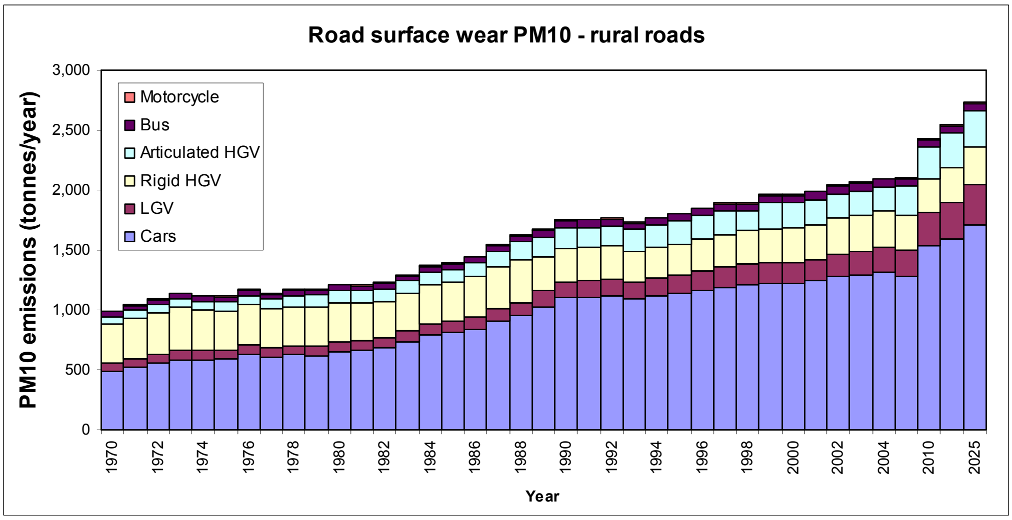

Figure I8: Total PM_{10} emissions due to road surface wear on urban roads in the UK (EMEP method).

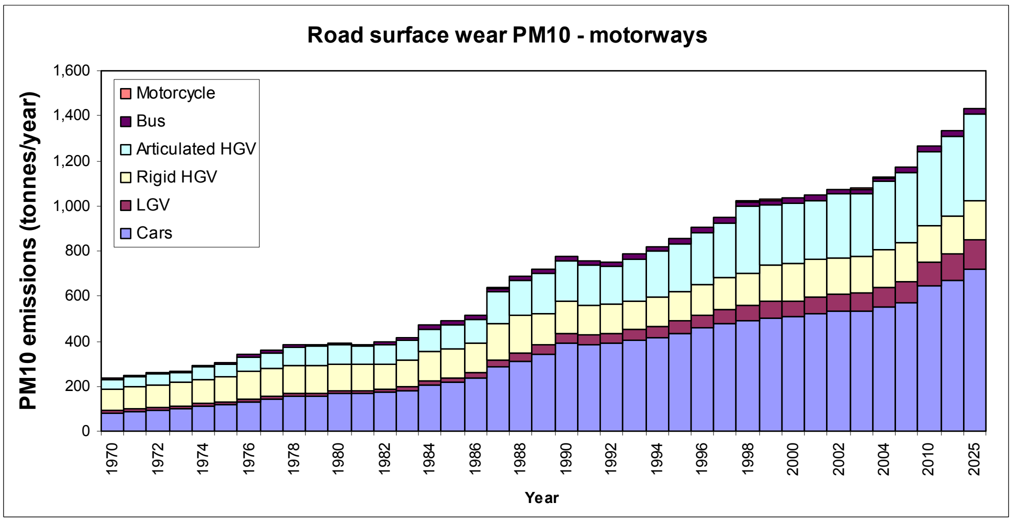

Figure I9: Total PM_{10} emissions due to road surface wear on urban roads in the UK (EMEP method).

Appendix J. Regional emission estimates – EMEP method

Table J1: PM_{10} emissions in England.

Vehicle category	Road type	Tyre wear (t/y) 2002	Tyre wear (t/y) 2003	Brake wear (t/y) 2002	Brake wear (t/y) 2003	Road wear (t/y) 2002	Road wear (t/y) 2003
Cars	Urban roads	1257.8	1245.5	1257.8	1245.5	1079.5	1068.9
	Rural Roads	890.9	899.8	890.9	899.8	982.6	992.4
	Motorways	368.0	367.2	368.0	367.2	476.6	475.6
LGVs	Urban roads	250.9	267.5	250.9	267.5	136.3	145.4
	Rural Roads	214.1	223.8	214.1	223.8	149.5	156.3
	Motorways	81.6	84.6	81.6	84.6	66.9	69.4
Rigid HGVs	Urban roads	69.2	71.8	69.2	71.8	126.8	131.6
	Rural Roads	99.2	99.4	99.2	99.4	217.7	218.0
	Motorways	52.8	52.9	52.8	52.9	143.6	143.8
Artic. HGVs	Urban roads	40.6	40.7	40.6	40.7	32.7	32.8
	Rural Roads	158.9	157.5	158.9	157.5	158.6	157.2
	Motorways	214.0	212.5	214.0	212.5	258.2	256.4
Buses	Urban roads	53.5	57.3	53.5	57.3	98.1	105.0
	Rural Roads	22.5	22.4	22.5	22.4	49.4	49.1
	Motorways	5.8	5.6	5.8	5.6	15.7	15.2
Motorcycles	Urban roads	9.1	10.6	9.1	10.6	7.2	8.5
	Rural Roads	5.2	5.3	5.2	5.3	5.3	5.5
	Motorways	0.9	1.0	0.9	1.0	1.1	1.1
Total		3794.9	3825.4	3794.9	3825.4	4005.8	4032.2

Table J2: $PM_{2.5}$ emissions in England.

Vehicle category	Road type	Tyre wear (t/y) 2002	Tyre wear (t/y) 2003	Brake wear (t/y) 2002	Brake wear (t/y) 2003	Road wear (t/y) 2002	Road wear (t/y) 2003
Cars	Urban roads	880.4	871.8	880.4	871.8	582.9	577.2
	Rural Roads	623.6	629.9	623.6	629.9	530.6	535.9
	Motorways	257.6	257.1	257.6	257.1	257.4	256.8
LGVs	Urban roads	175.6	187.3	175.6	187.3	73.6	78.5
	Rural Roads	149.9	156.7	149.9	156.7	80.7	84.4
	Motorways	57.1	59.2	57.1	59.2	36.1	37.5
Rigid HGVs	Urban roads	48.4	50.3	48.4	50.3	68.5	71.1
	Rural Roads	69.5	69.5	69.5	69.5	117.6	117.7
	Motorways	37.0	37.0	37.0	37.0	77.5	77.7
Artic. HGVs	Urban roads	28.4	28.5	28.4	28.5	17.7	17.7
	Rural Roads	111.2	110.3	111.2	110.3	85.6	84.9
	Motorways	149.8	148.7	149.8	148.7	139.4	138.4
Buses	Urban roads	37.5	40.1	37.5	40.1	53.0	56.7
	Rural Roads	15.7	15.6	15.7	15.6	26.7	26.5
	Motorways	4.0	3.9	4.0	3.9	8.5	8.2
Motorcycles	Urban roads	6.3	7.4	6.3	7.4	3.9	4.6
	Rural Roads	3.6	3.7	3.6	3.7	2.9	2.9
	Motorways	0.6	0.7	0.6	0.7	0.6	0.6
Total		2656.5	2677.8	2656.5	2677.8	2163.1	2177.4

Table J3: PM_1 emissions in England.

Vehicle category	Road type	Tyre wear (t/y) 2002	2003	Brake wear (t/y) 2002	2003	Road wear (t/y) 2002	2003
Cars	Urban roads	125.8	124.5	125.8	124.5	No EF	No EF
	Rural Roads	89.1	90.0	89.1	90.0	No EF	No EF
	Motorways	36.8	36.7	36.8	36.7	No EF	No EF
LGVs	Urban roads	25.1	26.8	25.1	26.8	No EF	No EF
	Rural Roads	21.4	22.4	21.4	22.4	No EF	No EF
	Motorways	8.2	8.5	8.2	8.5	No EF	No EF
Rigid HGVs	Urban roads	6.9	7.2	6.9	7.2	No EF	No EF
	Rural Roads	9.9	9.9	9.9	9.9	No EF	No EF
	Motorways	5.3	5.3	5.3	5.3	No EF	No EF
Artic. HGVs	Urban roads	4.1	4.1	4.1	4.1	No EF	No EF
	Rural Roads	15.9	15.8	15.9	15.8	No EF	No EF
	Motorways	21.4	21.2	21.4	21.2	No EF	No EF
Buses	Urban roads	5.4	5.7	5.4	5.7	No EF	No EF
	Rural Roads	2.2	2.2	2.2	2.2	No EF	No EF
	Motorways	0.6	0.6	0.6	0.6	No EF	No EF
Motorcycles	Urban roads	0.9	1.1	0.9	1.1	No EF	No EF
	Rural Roads	0.5	0.5	0.5	0.5	No EF	No EF
	Motorways	0.1	0.1	0.1	0.1	No EF	No EF
Total		379.5	382.5	379.5	382.5	No EF	No EF

Table J4: PM_{coarse} emissions in England.

Vehicle category	Road type	Tyre wear (t/y) 2002	2003	Brake wear (t/y) 2002	2003	Road wear (t/y) 2002	2003
Cars	Urban roads	377.3	373.6	377.3	373.6	496.6	491.7
	Rural Roads	267.3	269.9	267.3	269.9	452.0	456.5
	Motorways	110.4	110.2	110.4	110.2	219.2	218.8
LGVs	Urban roads	75.3	80.3	75.3	80.3	62.7	66.9
	Rural Roads	64.2	67.2	64.2	67.2	68.8	71.9
	Motorways	24.5	25.4	24.5	25.4	30.8	31.9
Rigid HGVs	Urban roads	20.8	21.5	20.8	21.5	58.3	60.5
	Rural Roads	29.8	29.8	29.8	29.8	100.2	100.3
	Motorways	15.9	15.9	15.9	15.9	66.1	66.2
Artic. HGVs	Urban roads	12.2	12.2	12.2	12.2	15.1	15.1
	Rural Roads	47.7	47.3	47.7	47.3	72.9	72.3
	Motorways	64.2	63.7	64.2	63.7	118.8	117.9
Buses	Urban roads	16.1	17.2	16.1	17.2	45.1	48.3
	Rural Roads	6.7	6.7	6.7	6.7	22.7	22.6
	Motorways	1.7	1.7	1.7	1.7	7.2	7.0
Motorcycles	Urban roads	2.7	3.2	2.7	3.2	3.3	3.9
	Rural Roads	1.6	1.6	1.6	1.6	2.5	2.5
	Motorways	0.3	0.3	0.3	0.3	0.5	0.5
Total		1138.5	1147.6	1138.5	1147.6	1842.7	1854.8

Table J5: PM_{10} emissions in Scotland.

Vehicle category	Road type	Tyre wear (t/y) 2002	Tyre wear (t/y) 2003	Brake wear (t/y) 2002	Brake wear (t/y) 2003	Road wear (t/y) 2002	Road wear (t/y) 2003
Cars	Urban roads	97.5	96.7	97.5	96.7	83.7	83.0
	Rural Roads	119.7	120.7	119.7	120.7	132.0	133.2
	Motorways	25.4	25.6	25.4	25.6	33.0	33.2
LGVs	Urban roads	18.9	20.2	18.9	20.2	10.3	11.0
	Rural Roads	31.0	32.4	31.0	32.4	21.6	22.6
	Motorways	5.3	5.5	5.3	5.5	4.3	4.5
Rigid HGVs	Urban roads	6.3	6.6	6.3	6.6	11.5	12.1
	Rural Roads	13.3	14.0	13.3	14.0	29.3	30.6
	Motorways	3.5	3.6	3.5	3.6	9.5	9.7
Artic. HGVs	Urban roads	3.7	3.7	3.7	3.7	3.0	3.0
	Rural Roads	21.4	22.1	21.4	22.1	21.3	22.1
	Motorways	14.2	14.3	14.2	14.3	17.1	17.3
Buses	Urban roads	6.3	6.7	6.3	6.7	11.6	12.2
	Rural Roads	4.8	4.9	4.8	4.9	10.6	10.7
	Motorways	0.6	0.6	0.6	0.6	1.7	1.6
Motorcycles	Urban roads	0.3	0.4	0.3	0.4	0.3	0.3
	Rural Roads	0.6	0.6	0.6	0.6	0.6	0.6
	Motorways	0.0	0.0	0.0	0.0	0.0	0.1
Total		372.9	378.6	372.9	378.6	401.4	407.7

Table J6: $PM_{2.5}$ emissions in Scotland.

Vehicle category	Road type	Tyre wear (t/y) 2002	Tyre wear (t/y) 2003	Brake wear (t/y) 2002	Brake wear (t/y) 2003	Road wear (t/y) 2002	Road wear (t/y) 2003
Cars	Urban roads	68.2	67.7	68.2	67.7	45.2	44.8
	Rural Roads	83.8	84.5	83.8	84.5	71.3	71.9
	Motorways	17.8	18.0	17.8	18.0	17.8	17.9
LGVs	Urban roads	13.2	14.2	13.2	14.2	5.5	5.9
	Rural Roads	21.7	22.7	21.7	22.7	11.7	12.2
	Motorways	3.7	3.8	3.7	3.8	2.3	2.4
Rigid HGVs	Urban roads	4.4	4.6	4.4	4.6	6.2	6.5
	Rural Roads	9.3	9.8	9.3	9.8	15.8	16.5
	Motorways	2.5	2.5	2.5	2.5	5.1	5.2
Artic. HGVs	Urban roads	2.6	2.6	2.6	2.6	1.6	1.6
	Rural Roads	15.0	15.5	15.0	15.5	11.5	11.9
	Motorways	9.9	10.0	9.9	10.0	9.3	9.3
Buses	Urban roads	4.4	4.7	4.4	4.7	6.3	6.6
	Rural Roads	3.4	3.4	3.4	3.4	5.7	5.8
	Motorways	0.4	0.4	0.4	0.4	0.9	0.9
Motorcycles	Urban roads	0.2	0.3	0.2	0.3	0.1	0.2
	Rural Roads	0.4	0.4	0.4	0.4	0.3	0.3
	Motorways	0.0	0.0	0.0	0.0	0.0	0.0
Total		261.0	265.0	261.0	265.0	216.8	220.2

Table J7: PM_1 emissions in Scotland.

Vehicle category	Road type	Tyre wear (t/y) 2002	2003	Brake wear (t/y) 2002	2003	Road wear (t/y) 2002	2003
Cars	Urban roads	9.7	9.7	9.7	9.7	No EF	No EF
	Rural Roads	12.0	12.1	12.0	12.1	No EF	No EF
	Motorways	2.5	2.6	2.5	2.6	No EF	No EF
LGVs	Urban roads	1.9	2.0	1.9	2.0	No EF	No EF
	Rural Roads	3.1	3.2	3.1	3.2	No EF	No EF
	Motorways	0.5	0.5	0.5	0.5	No EF	No EF
Rigid HGVs	Urban roads	0.6	0.7	0.6	0.7	No EF	No EF
	Rural Roads	1.3	1.4	1.3	1.4	No EF	No EF
	Motorways	0.4	0.4	0.4	0.4	No EF	No EF
Artic. HGVs	Urban roads	0.4	0.4	0.4	0.4	No EF	No EF
	Rural Roads	2.1	2.2	2.1	2.2	No EF	No EF
	Motorways	1.4	1.4	1.4	1.4	No EF	No EF
Buses	Urban roads	0.6	0.7	0.6	0.7	No EF	No EF
	Rural Roads	0.5	0.5	0.5	0.5	No EF	No EF
	Motorways	0.1	0.1	0.1	0.1	No EF	No EF
Motorcycles	Urban roads	0.0	0.0	0.0	0.0	No EF	No EF
	Rural Roads	0.1	0.1	0.1	0.1	No EF	No EF
	Motorways	0.0	0.0	0.0	0.0	No EF	No EF
Total		37.3	37.9	37.3	37.9	No EF	No EF

Table J8: PM_{coarse} emissions in Scotland.

Vehicle category	Road type	Tyre wear (t/y) 2002	2003	Brake wear (t/y) 2002	2003	Road wear (t/y) 2002	2003
Cars	Urban roads	29.2	29.0	29.2	29.0	38.5	38.2
	Rural Roads	35.9	36.2	35.9	36.2	60.7	61.3
	Motorways	7.6	7.7	7.6	7.7	15.2	15.3
LGVs	Urban roads	5.7	6.1	5.7	6.1	4.7	5.1
	Rural Roads	9.3	9.7	9.3	9.7	10.0	10.4
	Motorways	1.6	1.6	1.6	1.6	2.0	2.1
Rigid HGVs	Urban roads	1.9	2.0	1.9	2.0	5.3	5.6
	Rural Roads	4.0	4.2	4.0	4.2	13.5	14.1
	Motorways	1.1	1.1	1.1	1.1	4.4	4.5
Artic. HGVs	Urban roads	1.1	1.1	1.1	1.1	1.4	1.4
	Rural Roads	6.4	6.6	6.4	6.6	9.8	10.2
	Motorways	4.3	4.3	4.3	4.3	7.9	7.9
Buses	Urban roads	1.9	2.0	1.9	2.0	5.3	5.6
	Rural Roads	1.4	1.5	1.4	1.5	4.9	4.9
	Motorways	0.2	0.2	0.2	0.2	0.8	0.8
Motorcycles	Urban roads	0.1	0.1	0.1	0.1	0.1	0.1
	Rural Roads	0.2	0.2	0.2	0.2	0.3	0.3
	Motorways	0.0	0.0	0.0	0.0	0.0	0.0
Total		111.9	113.6	111.9	113.6	184.7	187.6

Table J9: PM_{10} emissions in Wales.

Vehicle category	Road type	Tyre wear (t/y) 2002	2003	Brake wear (t/y) 2002	2003	Road wear (t/y) 2002	2003
Cars	Urban roads	62.5	62.2	62.5	62.2	53.7	53.4
	Rural Roads	79.8	81.2	79.8	81.2	88.1	89.6
	Motorways	13.8	14.1	13.8	14.1	17.9	18.3
LGVs	Urban roads	11.4	12.1	11.4	12.1	6.2	6.6
	Rural Roads	21.2	22.2	21.2	22.2	14.8	15.5
	Motorways	3.1	3.3	3.1	3.3	2.6	2.7
Rigid HGVs	Urban roads	3.2	3.3	3.2	3.3	5.9	6.1
	Rural Roads	7.6	7.5	7.6	7.5	16.6	16.5
	Motorways	1.5	1.5	1.5	1.5	4.2	4.0
Artic. HGVs	Urban roads	1.9	1.9	1.9	1.9	1.5	1.5
	Rural Roads	12.1	11.9	12.1	11.9	12.1	11.9
	Motorways	6.3	6.0	6.3	6.0	7.5	7.2
Buses	Urban roads	2.3	2.5	2.3	2.5	4.3	4.6
	Rural Roads	2.7	2.7	2.7	2.7	5.9	6.0
	Motorways	0.2	0.2	0.2	0.2	0.5	0.5
Motorcycles	Urban roads	0.3	0.3	0.3	0.3	0.2	0.3
	Rural Roads	0.4	0.4	0.4	0.4	0.4	0.4
	Motorways	0.0	0.0	0.0	0.0	0.0	0.0
Total		230.4	233.4	230.4	233.4	242.4	245.0

Table J10: $PM_{2.5}$ emissions in Wales.

Vehicle category	Road type	Tyre wear (t/y) 2002	2003	Brake wear (t/y) 2002	2003	Road wear (t/y) 2002	2003
Cars	Urban roads	43.8	43.6	43.8	43.6	29.0	28.8
	Rural Roads	55.9	56.9	55.9	56.9	47.6	48.4
	Motorways	9.7	9.9	9.7	9.9	9.7	9.9
LGVs	Urban roads	8.0	8.5	8.0	8.5	3.3	3.5
	Rural Roads	14.8	15.5	14.8	15.5	8.0	8.4
	Motorways	2.2	2.3	2.2	2.3	1.4	1.5
Rigid HGVs	Urban roads	2.2	2.3	2.2	2.3	3.2	3.3
	Rural Roads	5.3	5.3	5.3	5.3	9.0	8.9
	Motorways	1.1	1.0	1.1	1.0	2.3	2.2
Artic. HGVs	Urban roads	1.3	1.3	1.3	1.3	0.8	0.8
	Rural Roads	8.5	8.3	8.5	8.3	6.5	6.4
	Motorways	4.4	4.2	4.4	4.2	4.1	3.9
Buses	Urban roads	1.6	1.7	1.6	1.7	2.3	2.5
	Rural Roads	1.9	1.9	1.9	1.9	3.2	3.2
	Motorways	0.1	0.1	0.1	0.1	0.3	0.3
Motorcycles	Urban roads	0.2	0.2	0.2	0.2	0.1	0.1
	Rural Roads	0.3	0.3	0.3	0.3	0.2	0.2
	Motorways	0.0	0.0	0.0	0.0	0.0	0.0
Total		161.3	163.4	161.3	163.4	130.9	132.3

Table J11: PM_1 emissions in Wales.

Vehicle category	Road type	Tyre wear (t/y) 2002	Tyre wear (t/y) 2003	Brake wear (t/y) 2002	Brake wear (t/y) 2003	Road wear (t/y) 2002	Road wear (t/y) 2003
Cars	Urban roads	6.3	6.2	6.3	6.2	No EF	No EF
	Rural Roads	8.0	8.1	8.0	8.1	No EF	No EF
	Motorways	1.4	1.4	1.4	1.4	No EF	No EF
LGVs	Urban roads	1.1	1.2	1.1	1.2	No EF	No EF
	Rural Roads	2.1	2.2	2.1	2.2	No EF	No EF
	Motorways	0.3	0.3	0.3	0.3	No EF	No EF
Rigid HGVs	Urban roads	0.3	0.3	0.3	0.3	No EF	No EF
	Rural Roads	0.8	0.8	0.8	0.8	No EF	No EF
	Motorways	0.2	0.1	0.2	0.1	No EF	No EF
Artic. HGVs	Urban roads	0.2	0.2	0.2	0.2	No EF	No EF
	Rural Roads	1.2	1.2	1.2	1.2	No EF	No EF
	Motorways	0.6	0.6	0.6	0.6	No EF	No EF
Buses	Urban roads	0.2	0.2	0.2	0.2	No EF	No EF
	Rural Roads	0.3	0.3	0.3	0.3	No EF	No EF
	Motorways	0.0	0.0	0.0	0.0	No EF	No EF
Motorcycles	Urban roads	0.0	0.0	0.0	0.0	No EF	No EF
	Rural Roads	0.0	0.0	0.0	0.0	No EF	No EF
	Motorways	0.0	0.0	0.0	0.0	No EF	No EF
Total		23.0	23.3	23.0	23.3	No EF	No EF

Table J12: PM_{coarse} emissions in Wales.

Vehicle category	Road type	Tyre wear (t/y) 2002	Tyre wear (t/y) 2003	Brake wear (t/y) 2002	Brake wear (t/y) 2003	Road wear (t/y) 2002	Road wear (t/y) 2003
Cars	Urban roads	18.8	18.7	18.8	18.7	24.7	24.6
	Rural Roads	24.0	24.4	24.0	24.4	40.5	41.2
	Motorways	4.2	4.2	4.2	4.2	8.2	8.4
LGVs	Urban roads	3.4	3.6	3.4	3.6	2.8	3.0
	Rural Roads	6.4	6.7	6.4	6.7	6.8	7.1
	Motorways	0.9	1.0	0.9	1.0	1.2	1.2
Rigid HGVs	Urban roads	1.0	1.0	1.0	1.0	2.7	2.8
	Rural Roads	2.3	2.3	2.3	2.3	7.6	7.6
	Motorways	0.5	0.4	0.5	0.4	1.9	1.9
Artic. HGVs	Urban roads	0.6	0.6	0.6	0.6	0.7	0.7
	Rural Roads	3.6	3.6	3.6	3.6	5.6	5.5
	Motorways	1.9	1.8	1.9	1.8	3.5	3.3
Buses	Urban roads	0.7	0.7	0.7	0.7	2.0	2.1
	Rural Roads	0.8	0.8	0.8	0.8	2.7	2.7
	Motorways	0.1	0.1	0.1	0.1	0.2	0.2
Motorcycles	Urban roads	0.1	0.1	0.1	0.1	0.1	0.1
	Rural Roads	0.1	0.1	0.1	0.1	0.2	0.2
	Motorways	0.0	0.0	0.0	0.0	0.0	0.0
Total		69.1	70.0	69.1	70.0	111.5	112.7

Table J13: PM_{10} emissions in Northern Ireland.

Vehicle category	Road type	Tyre wear (t/y) 2002	Tyre wear (t/y) 2003	Brake wear (t/y) 2002	Brake wear (t/y) 2003	Road wear (t/y) 2002	Road wear (t/y) 2003
Cars	Urban roads	36.6	37.5	36.6	37.5	31.4	32.2
	Rural Roads	66.7	68.4	66.7	68.4	73.6	75.4
	Motorways	5.9	6.1	5.9	6.1	7.7	7.9
LGVs	Urban roads	6.3	7.0	6.3	7.0	3.4	3.8
	Rural Roads	11.4	12.8	11.4	12.8	8.0	8.9
	Motorways	1.0	1.1	1.0	1.1	0.8	0.9
Rigid HGVs	Urban roads	6.7	7.0	6.7	7.0	12.4	12.8
	Rural Roads	15.5	16.0	15.5	16.0	34.0	35.1
	Motorways	1.8	1.9	1.8	1.9	4.9	5.1
Artic. HGVs	Urban roads	2.2	2.3	2.2	2.3	1.8	1.9
	Rural Roads	6.3	6.5	6.3	6.5	6.3	6.5
	Motorways	1.5	1.6	1.5	1.6	1.8	1.9
Buses	Urban roads	0.7	0.7	0.7	0.7	1.2	1.3
	Rural Roads	0.5	0.6	0.5	0.6	1.2	1.2
	Motorways	0.0	0.0	0.0	0.0	0.1	0.1
Motorcycles	Urban roads	0.2	0.3	0.2	0.3	0.2	0.2
	Rural Roads	0.1	0.2	0.1	0.2	0.1	0.2
	Motorways	0.0	0.0	0.0	0.0	0.0	0.0
Total		163.6	170.0	163.6	170.0	188.9	195.5

Table J14: $PM_{2.5}$ emissions in Northern Ireland.

Vehicle category	Road type	Tyre wear (t/y) 2002	Tyre wear (t/y) 2003	Brake wear (t/y) 2002	Brake wear (t/y) 2003	Road wear (t/y) 2002	Road wear (t/y) 2003
Cars	Urban roads	25.6	26.3	25.6	26.3	17.0	17.4
	Rural Roads	46.7	47.9	46.7	47.9	39.7	40.7
	Motorways	4.2	4.3	4.2	4.3	4.2	4.3
LGVs	Urban roads	4.4	4.9	4.4	4.9	1.8	2.1
	Rural Roads	8.0	9.0	8.0	9.0	4.3	4.8
	Motorways	0.7	0.8	0.7	0.8	0.4	0.5
Rigid HGVs	Urban roads	4.7	4.9	4.7	4.9	6.7	6.9
	Rural Roads	10.8	11.2	10.8	11.2	18.3	19.0
	Motorways	1.3	1.3	1.3	1.3	2.6	2.7
Artic. HGVs	Urban roads	1.6	1.6	1.6	1.6	1.0	1.0
	Rural Roads	4.4	4.6	4.4	4.6	3.4	3.5
	Motorways	1.1	1.1	1.1	1.1	1.0	1.0
Buses	Urban roads	0.5	0.5	0.5	0.5	0.7	0.7
	Rural Roads	0.4	0.4	0.4	0.4	0.6	0.7
	Motorways	0.0	0.0	0.0	0.0	0.1	0.1
Motorcycles	Urban roads	0.1	0.2	0.1	0.2	0.1	0.1
	Rural Roads	0.1	0.1	0.1	0.1	0.1	0.1
	Motorways	0.0	0.0	0.0	0.0	0.0	0.0
Total		114.6	119.0	114.6	119.0	102.0	105.6

Table J15: PM_1 emissions in Northern Ireland.

Vehicle category	Road type	Tyre wear (t/y) 2002	2003	Brake wear (t/y) 2002	2003	Road wear (t/y) 2002	2003
Cars	Urban roads	3.7	3.8	3.7	3.8	No EF	No EF
	Rural Roads	6.7	6.8	6.7	6.8	No EF	No EF
	Motorways	0.6	0.6	0.6	0.6	No EF	No EF
LGVs	Urban roads	0.6	0.7	0.6	0.7	No EF	No EF
	Rural Roads	1.1	1.3	1.1	1.3	No EF	No EF
	Motorways	0.1	0.1	0.1	0.1	No EF	No EF
Rigid HGVs	Urban roads	0.7	0.7	0.7	0.7	No EF	No EF
	Rural Roads	1.5	1.6	1.5	1.6	No EF	No EF
	Motorways	0.2	0.2	0.2	0.2	No EF	No EF
Artic. HGVs	Urban roads	0.2	0.2	0.2	0.2	No EF	No EF
	Rural Roads	0.6	0.7	0.6	0.7	No EF	No EF
	Motorways	0.2	0.2	0.2	0.2	No EF	No EF
Buses	Urban roads	0.1	0.1	0.1	0.1	No EF	No EF
	Rural Roads	0.1	0.1	0.1	0.1	No EF	No EF
	Motorways	0.0	0.0	0.0	0.0	No EF	No EF
Motorcycles	Urban roads	0.0	0.0	0.0	0.0	No EF	No EF
	Rural Roads	0.0	0.0	0.0	0.0	No EF	No EF
	Motorways	0.0	0.0	0.0	0.0	No EF	No EF
Total		16.4	17.0	16.4	17.0	No EF	No EF

Table J16: PM_{coarse} emissions in Northern Ireland.

Vehicle category	Road type	Tyre wear (t/y) 2002	2003	Brake wear (t/y) 2002	2003	Road wear (t/y) 2002	2003
Cars	Urban roads	11.0	11.3	11.0	11.3	14.5	14.8
	Rural Roads	20.0	20.5	20.0	20.5	33.9	34.7
	Motorways	1.8	1.8	1.8	1.8	3.5	3.6
LGVs	Urban roads	1.9	2.1	1.9	2.1	1.6	1.8
	Rural Roads	3.4	3.8	3.4	3.8	3.7	4.1
	Motorways	0.3	0.3	0.3	0.3	0.4	0.4
Rigid HGVs	Urban roads	2.0	2.1	2.0	2.1	5.7	5.9
	Rural Roads	4.6	4.8	4.6	4.8	15.6	16.2
	Motorways	0.5	0.6	0.5	0.6	2.3	2.3
Artic. HGVs	Urban roads	0.7	0.7	0.7	0.7	0.8	0.9
	Rural Roads	1.9	2.0	1.9	2.0	2.9	3.0
	Motorways	0.5	0.5	0.5	0.5	0.8	0.9
Buses	Urban roads	0.2	0.2	0.2	0.2	0.6	0.6
	Rural Roads	0.2	0.2	0.2	0.2	0.5	0.6
	Motorways	0.0	0.0	0.0	0.0	0.0	0.0
Motorcycles	Urban roads	0.1	0.1	0.1	0.1	0.1	0.1
	Rural Roads	0.0	0.1	0.0	0.1	0.1	0.1
	Motorways	0.0	0.0	0.0	0.0	0.0	0.0
Total		49.1	51.0	49.1	51.0	86.9	89.9