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Evaluation of a road dust suspension model for predicting the concentrations of PM_{10} in a street canyon

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A B S T R A C T

We have slightly refined, evaluated and tested a mathematical model for predicting the vehicular suspension emissions of PM_{10}. The model describes particulate matter generated by the wear of road pavement, traction sand, and the processes that control the suspension of road dust particles into the air. However, the model does not address the emissions from the wear of vehicle components. The performance of this suspension emission model has been evaluated in combination with the street canyon dispersion model OSPM. We used data from a measurement campaign that was conducted in the street canyon Runeberg Street in Helsinki from 8 January to 2 May, 2004. The model reproduced fairly well the seasonal variation of the PM_{10} concentrations, also during the time periods, when studded tyres and anti-skid treatments were commonly in use. For instance, the index of agreement (IA) was 0.83 for the time series of the hourly predicted and observed concentrations of PM_{10}. The predictions of the model were found to be sensitive to precipitation and street traction sanding. The main uncertainties in the predictions are probably caused by (i) the cleaning processes of the streets, which are currently not included in the model, (ii) the uncertainties in the estimation of the sanding days, and (iii) the uncertainties in the evaluation of precipitation. This study provides more confidence that this model could potentially be a valuable tool of assessment to evaluate and forecast the suspension PM_{10} emissions worldwide. However, a further evaluation of the model is needed against other datasets in various vehicle fleet, speed and climatic conditions.

1. Introduction

The vehicular non-exhaust particles are generated mechanically from the wear of vehicle components (such as brakes, clutches and tyres) and road surfaces (e.g., Kupiainen, 2007; Ketzel et al., 2007), or suspended by wind and vehicle-induced turbulence. In Scandinavia, Japan and several states ofthe USA, local measures, such as reducing the share of cars with studded tyres, have already resulted in reductions of non-exhaust emissions (e.g., Ketzel et al., 2007; Johansson, 2008). In order to carry out efficient traffic planning and air quality management, validated modelling tools are needed that include also non-exhaust particulate matter (e.g., Keuken, 2006).

The US-EPA AP-42 model was one of the first non-exhaust vehicle emission models. It is based on the average silt load of the road and the average weight of the vehicle fleet (US-EPA, 2003). The modified US-EPA model used in Germany, e.g., by Düring et al. (2004) considers also, e.g., the type of the street, the number of days with rain, and the quality of the street surface. However, the AP-42 model has been criticized for lacking the mechanistic basis and being highly dependent on the used data set. The silt load also cannot be measured accurately (Venkatram, 2000; Düring et al., 2004). In Germany, the procedure of Gehrig et al. (2004) is therefore currently used for non-exhaust emission factors (Düring et al., 2004).

Gehrig et al. (2004) have derived emission factors of PM_{10} and PM_1 based on concentration differences of NO_x and particulate matter for different traffic situations. They considered PM_1 as exhaust emissions and the difference of PM_{10}–PM_1 as non-exhaust emissions. Similarly, Thorpe et al. (2007) have used particle and NO_x concentrations to estimate emission factors of on-road particle suspension. The total source strength of coarse particles ($PM_{2.5-10}$) was estimated by subtracting the background concentrations from

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the roadside concentrations and converting this to source strength by its ratio to NO_x and assuming an estimated NO_x emission factor.

Traffic related non-exhaust particle emissions have been determined also by source apportionment studies. Abu-Allaban et al. (2003) have derived emission factors from measurements of particles in the United States, by applying chemical mass balance receptor modelling to the SEM-analysed data. Amato et al. (2009) and Bukowiecki et al. (2010) have applied positive matrix factorization in determination of the contribution of non-exhaust particles in Spain and Switzerland, respectively.

Kukkonen et al. (2001a) developed a model for evaluating the long term average spatial concentration distributions of PM_{10}, based on the corresponding modelled NO_x concentrations. The model assumption was that local vehicular traffic is responsible for a substantial fraction of the street-level concentrations of both PM_{10} and NO_x, either due to primary emissions or suspension.

In Nordic countries, where street sanding and studded tyres may be used in winter, the particle emissions from the road and tyre wear commonly correlate seasonally poorly with exhaust emissions. Tønnesen (2005) presented a model for roadside PM_{10} emissions that assumed linear relations between the share of heavy duty vehicles and suspended particles. The model considered also the average driving speed, the share of studded tyres, and the road surface conditions. However, the model lacks the phenomenon where the dust, built up during the winter due to the wet or snowy street surfaces, is released after it becomes sufficiently dry.

Omstedt et al. (2005) accounted the dust layer build up during wet conditions by the wear of road pavement and traction sand, and its reduction during dry conditions by the suspension of road dust particles into the air. The surface emissions are estimated by taking into account precipitation, evaporation, and water runoff. The model predicts suspension emissions in varying meteorological conditions. This model could potentially be a valuable tool of assessment to evaluate the suspension PM_{10} emissions worldwide. However, the evaluation studies are very scarce at the moment; published studies have been conducted only in Sweden and Denmark up to date. The model can also only be used as a research tool for diagnostic purposes, but not for air quality forecasting.

The first aim of this study is to present a slightly refined version of the original model (Omstedt et al., 2005) that is applicable also in operational forecasting of air quality. We will also present the model equations in a more rigorous form, and discuss the mathematical and numerical differences between the original and refined models. The second aim of this study is to evaluate the refined model in combination with the street canyon dispersion model OSPM (e.g., Berkowicz, 2000), against a measurement campaign in a street canyon (Runeberg Street) in Helsinki, in 2004. This will provide an independent critical evaluation of the model in another country. The third aim of this paper is to study the sensitivity of the refined model with respect to various key factors, including weather conditions and street sanding, in order to obtain a better insight on the model behaviour.

2. Materials and methods

2.1. Measurement

The studied street segment of the Runeberg Street is approximately north–south oriented (6°–186°), 24 m wide, and the height of the buildings on both sides of the street is 23 m (Fig. 1a–b). The building structure on the western side is uniform over a distance of 175 m. On the eastern side, there is one small junction at about 38 m from the measurement site to the north (Kukkonen et al., 2001b). The street level PM_{10} concentrations were measured at the height of 4.5 m above the pavement. The urban background air

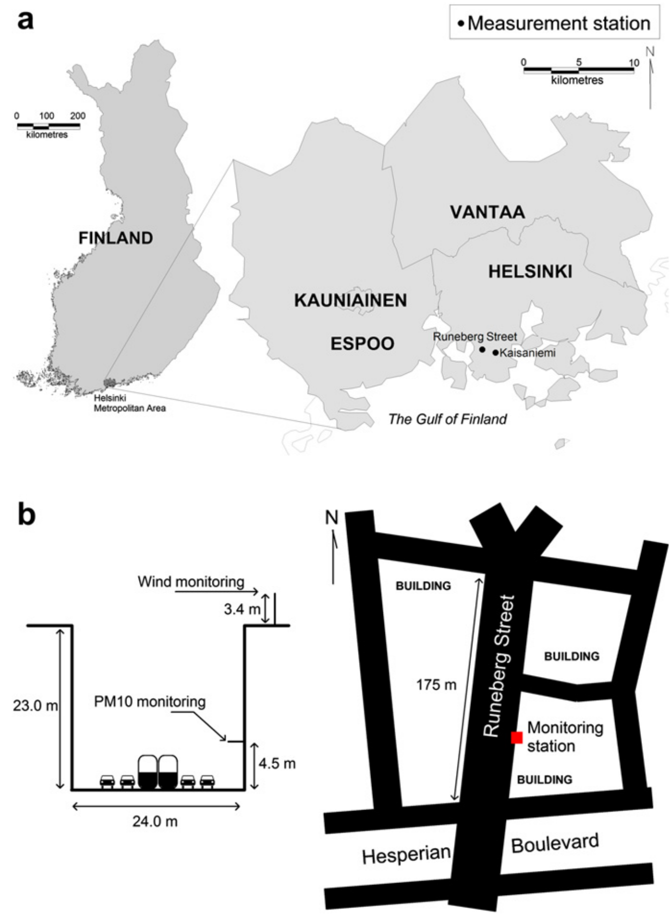

Fig. 1. a–b. The locations of the street canyon site Runeberg Street and urban background site Kaisaniemi in Helsinki, and locations of the measurement points in Runeberg Street.

quality station of Kaisaniemi was located in the Helsinki city centre about 1.4 km south-east from the Runeberg Street. The PM_{10} concentrations were measured at a height of 2 m on a roof of 20 m high building situated on the edge of a park.

The wind speed and direction were monitored at a height of 3.4 m on a roof of a 23 m high building at the Runeberg Street. Temperature and relative humidity were measured at Kaisaniemi at a height of 2.0 m and precipitation at the height of 1.5 m. A summary of monitoring instruments is presented in Table 1. The monthly share of traffic volume and speed, as well as, the fleet composition was measured by an automatic monitoring site at Runeberg Street next to the concentration measurement site. Identification of vehicles was based on two consecutive induction loops. The measurements at Runeberg Street and Kaisaniemi were conducted from 8 January to 2 May in 2004.

Table 1
The measured quantities and the observational methods.

Measured quantity	Dimension	Measuring principle	Instrument
PM_{10} concentration	$\mu g\ m^{-3}$	β-attenuation	Eberline FH 62 I-R
Wind speed	$m\ s^{-1}$	anemometer	Vaisala WAA151
Wind direction	degrees	ultrasounding	Vaisala WAS425AH
Precipitation	$mm\ h^{-1}$	gravimetric	Vaisala RG13H
Relative humidity	%	capacitive polymer sensor	Vaisala HMP35
Global radiation	$W\ m^{-2}$	pyranometer	Kipp & Zonen CM11
Cloud cover	octas	ceilometer	Vaisala CT25K
Temperature	°C	resistive detector	Pentronic Pt-100