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Source apportionment of particulate matter using dispersion and receptor modelling

A case study for Oslo

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1 Executive Summary

Emission inventories for air pollution sources are in many cases the basis for air quality assessment and management in European cities. The spatial and temporal distribution of the emission sources is estimated as accurate as possible in the inventories. In practice, atmospheric emissions are estimated on the basis of measurements made at selected or representative samples of the sources and source types. The quality of the inventories is of great importance since these data are used for air quality assessment and management of air pollution problems. As a part of the EU-funded project Air4EU, a source apportionment study has been performed for Oslo. In this case study, the quality of the emission inventory for particulates ($PM_{2.5}$) was assessed. This assessment was performed by comparing source apportionment estimates from a dispersion model, using the official emission inventory, with source apportionment estimates from a receptor model. This document provides an example of a recommended method for carrying out such an assessment.

The source apportionment study is based on data from a measurement campaign performed from January 2004 until end of April 2004 and from October 2004 until end of April 2005. For this campaign, NILU analysed the chemical composition of $PM_{2.5}$ on filter samples. With these data as input, Positive Matrix Factorization (PMF) receptor model was applied to identify and quantify the various source contributions. For the same observational period and site, we performed emission and dispersion modelling using the Air Quality Management system AirQUIS (www.airquis.com).

The assessment of emission inventory for $PM_{2.5}$ was performed by comparing the estimated source strengths for the different source categories from the PMF model with the dispersion model. The PMF analysis was performed on 78 filter samples (12-hour samples) for interpretation of the source categories. The comparisons of the source strengths for the different sources with the dispersion model were only performed for the 40 filters collected at Rv4 from January until end of April 2004.

The results identified gaps and weaknesses in the dispersion model and in the receptor model results. In comparison to the receptor model, the dispersion model overestimated contributions due to wood burning by a factor of 2.0 and underestimated contributions from re-suspension by a factor of 6.7. However, on the average the deviation between the estimated and measured $PM_{2.5}$ was only 13%.

To assess if the differences between dispersion and receptor model calculations, found at the one site, were generally applicable throughout Oslo, we adjusted the emission inventory for individual source categories by a simple rescaling of the emission rates and recalculated $PM_{2.5}$ concentrations using the dispersion model. These adjusted $PM_{2.5}$ concentrations were compared with measurements at three other independent stations to evaluate the improvements of the updated inventory. The statistical analyses of these results showed an improvement in the dispersion estimated for $PM_{2.5}$ for all the stations.

2 Case study description

2.1 Background

According to the EU guidelines (EC, 1999) national and local authorities are required to provide an assessment of the air quality levels. For these impact assessment studies, the official emission inventories are used for dispersion and exposure calculations. In addition, source contribution must be calculated if the limit values and guidelines are exceeded, since abatement measures must be implemented. For this reason it is not only important to calculate the concentration levels properly, but also the source apportionment of the pollutant to identify the correct abatement measures.

During the winter and spring seasons, Norwegian cities are susceptible to poor air quality events that can lead to concentrations in exceedence of EU limit values. Such events typically take place during periods with strong temperature inversions, weak winds and little vertical mixing. For particulate matter, these episodes are enhanced during cold and dry conditions when emission of PM from domestic wood burning and from traffic induced re-suspension are at their highest. The problem of PM emission by traffic re-suspension is compounded in Scandinavian cities by the use of studded tyres, which produce a large reservoir of dust particles during their winter time use as well as enhanced emission from impact with road surfaces and re-suspension due to vehicle turbulence.

In Oslo the emission inventory for industrial and area distributed sources has been compiled by Statistics Norway. The traffic emission inventory is based on official data from The Norwegian Public Roads Administration.

The emission inventory is a fundamental basis for air pollution impact assessments using dispersion modelling. The quality of the assessment is highly dependent on the quality of the input data, as well as other factors. Methods of independent evaluations and validation of inventories/dispersion results should be addressed. An independent assessment of the emission inventory can be performed using receptor modelling.

2.2 Aim and description

The aim of this case study is to provide an example of a methodology that can be used to perform an independent assessment of emission inventories for PM, and how the methodology can be used to empirically improve emission estimates and resulting air quality estimates of PM in urban areas. Comparing source strengths for different source categories estimated using dispersion and receptor modelling performs the assessment of the emission inventory for $PM_{2.5}$. The dispersion model uses the emission inventory as input for the source apportionment estimates and the receptor model uses data from the measurement campaign. Methodologies for assessing uncertainties of model estimates are applied to identify the overall ability of the model to predict the observed values both before and after the emission inventory has been updated to identify any improvements in the model estimates.

This study is based on data from a measurement campaign performed from January 2004 until end of April 2005 in Oslo (Hagen et al., 2005). For this campaign the chemical composition of $PM_{2.5}$ from filter samples collected at a road side station, Rv4, was analysed. With these data as input, Positive Matrix Factorization (PMF) receptor modelling was applied to detect and quantify the various source contributions. For the same observational period and site, we performed emission and dispersion model calculations using the Air Quality Management system AirQUIS (www.airquis.com).

2.3 Relevance to recommendations in Air4EU

This case study evaluates whether receptor models can be used for the independent assessment of emission inventories on the urban scale. The results are relevant for the assessment of source contributions to PM. This case study supports recommendations concerning the use of such methods for evaluation and improvement of the quality of emission inventories used as input for air quality assessment and source contributions to PM. The methodology applied here, using intensive measurement analysis and receptor modelling, must be considered as an advanced 'best practice' or research level activity. In other case studies carried out within Air4EU (Air4EU – CS D7.1.1 and D7.1.2) simpler analysis methods are recommended based on more normal operational measurements.

3 Methodology

3.1 Measurements of $PM_{2.5}$

In 2004, the permanent air quality monitoring network in Oslo consisted of 10 monitoring stations, five street stations and five urban background stations. At five of these stations $PM_{2.5}$ were measured using TEOM instruments (Lützenkirchen and Lutnæs, 2004).

In parallel and for two different periods, from January 2004 until end of April 2004 and from October 2004 until end of April 2005, a measurement campaign was carried at the same site as one of the permanent stations, Rv4, in Oslo. 12 hour filter samples of $PM_{2.5}$ were collected using Kleinfiltergerat (KFG) instrument (Hagen et al., 2005). A selected sample of these filters was analysed for chemical composition at NILU.

Measurements of hourly averages of $PM_{2.5}$ are available from January to end of April 2004 at four of the stations in Oslo. These stations, Kirkeveien, Løren, Aker hospital in addition to Rv4 (Figure 1), are used to independently evaluate the impact of modifying the emission inventory for the model domain based on the source apportionment analysis at Rv4. Kirkeveien is dominated by traffic/wood burning, Løren and Rv4 are traffic stations and Aker hospital is an urban background station.

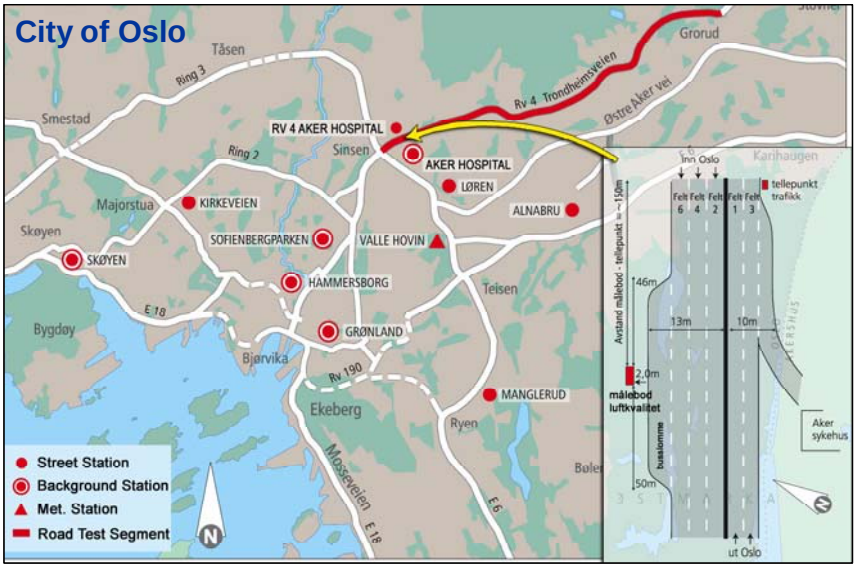

Figure 1: Map showing the model area and the location of monitoring stations in Oslo and a detailed description of the location of the Rv4 station.

In addition to the local monitoring stations regional background concentrations for $PM_{2.5}$, used in the dispersion model calculations, are available from the EMEP station Birkenes, located approximately 300 km from Oslo in the south of Norway (58º 23'N, 8º 15'E). Daily values of $PM_{2.5}$ have been used as input to the dispersion model. Samples are collected using a Rupprecht & Patashnick Dichotomous Partisol-Plus. The particulate mass for the two fractions is determined gravimetrically.

3.2 Chemical Analysis of the PM filters

78 $PM_{2.5}$ filter samples were selected for chemical analysis (Hagen et al., 2005). The components and elements analysed were selected to provide as much information as possible for the source identification. The filter samples were analysed with respect to major anions and cations using ion chromatography, and 30 element using ICP-MS (Inductively Coupled Plasma Mass Spectroscopy) (NILU, 2006).

Thermal Optical analysis (TOT) was used to quantify the samples content of elemental (EC) and organic carbon (OC). (Birch and Cary, 1996), whereas levoglucosan (Dye et al., 2005; Yttri et al., 2005) and NCBA where quantified using HPLC/HRMS (High Performance Liquid Chromatography /High-Resolution Mass Spectrometry)

3.3 Emission inventory and emission calculations

The emission inventory for Oslo is a bottom-up inventory and is compiled based on a detailed knowledge of source types, their locations, and their specific emissions or consumption data. The emission inventory includes all types of emission data, which are pooled, for dispersion modelling purpose, into three types of sources, namely point, area and line sources.

Emission from traffic is introduced into the model in two separate ways, as area or as line source emissions, depending on the average daily traffic intensity. Receptor points close to roads, where most monitoring stations are located, are calculated using the line source model (HIWAY-2) for sources closer than 500 m and the Eulerian model for sources further away.

The emission model for traffic-induced re-suspension of PM is dependent upon traffic speed, heavy-duty vehicle fraction, percentage of studded tyres and road surface condition (Tønnesen, 2005).

The emission equation for re-suspension for $PM_{2.5}$, is a function of percentage of studded tyres, road surface condition and on exhaust particle emission factor. It is expressed in the following way (Tønnesen, 2005):

$$Q_{PM_{2.5}\ fraction} = Q_{EP} + Q_{resusp}$$

where

$$Q_{PM_{2.5}\ fraction} = Q_{EP} + Q_{EP} * 0.69 * RP * RW$$

QEP	= Calculated emission from Exhaust Particles
RP	= Re-suspension Factor = 0.98*Percentage of studded tyre + 0.02
RW	= Reduction Factor due to wet road. Default is 1 if no meteorology data exist.

The reduction factor due to the road surface condition takes into account precipitation and hours since last rainfall, air temperature, time of the day, surface temperature and dew point temperature if available (Tønnesen, 2003)

The above empirical equation for $PM_{2.5}$ assumes that re-suspension is dependent on the exhaust particle level and decreases with lower vehicle emissions. The traffic emission inventory is based on official data from The Norwegian Public Roads Administration. The emissions factors for vehicles are from “Emission from road traffic in Norway” (Bang et al., 1999), Traffic emission regulations (Statens vegvesen Vegdirektoratet, 2002) and Copert III (Ntziachristos and Samaras, 2000, Kouridis et al., 2000).

During the winter 2003/2004, approximately 30 % of the vehicles in Oslo used studded tyres.

Emission and consumption data for both area sources as well as industrial sources are provided by Statistics Norway and are valid for 1998. An exception from this is the emission inventory for wood burning, which is based on a survey of use of wood for heating, and heating habits in Oslo ("The Oslo survey") carried out in the autumn 2002 (Finstad et al., 2004). The data for total wood consumption, the spatial and temporal distribution for wood burning used in the calculations is from September 2001 until end of August 2002. The total emissions are distributed on weeks, days and hours. The daily and hourly time variations are based on people's living habits. The weekly time variation is based on the 30 year temperature average from 1961 to 1990.

Emissions of PM from wood burning and other area distributed emission sources, except traffic, are introduced into the model as area sources into the three lowest levels of the Eulerian model (71 m for Oslo).

3.4 Dispersion modelling

The AirQUIS modelling system developed by NILU is applied in this study to calculate concentrations of particulates (AirQUIS, 2006). AirQUIS is a GIS based integrated management system that includes a user interface, comprehensive measurement and emission inventory databases, and a suite of models for use in simulating ambient air concentrations and exposure.

The models used in the calculations are the MATHEW diagnostic wind field model (Sherman, 1978; Foster et al., 1995) and the EPISODE dispersion model (Slørdal et al., 2003). The dispersion model contains a Eulerian model with embedded sub-grid line source and point source models for calculating ambient concentrations. The line source model HIWAY-2 (Petersen, 1980) is used to calculate traffic related contributions at receptor points close to roads.

Boundary concentrations for the model area are measurements from the regional background station Birkenes (section 3.1).

The grid applied for the Oslo region is a 22 x 18 km grid, grid size 1 km, with 10 vertical levels ranging up to 2400 m. The meteorological field is calculated with MATHEW using input from a meteorological mast situated at Valle Hovin (Figure 1). Calculations are carried out for a winter-spring period, from November 2003 up to April 2004.

Concentrations calculated at receptor points corresponding to the positions of monitoring stations in Oslo are recorded for comparison with monitoring data and the filter samples (KFG) at Rv4.

To carry out the source apportionment using the dispersion model, calculations of the hourly concentrations from single source categories or a suite of categories were performed. The classification of sources was made according to the identified sources from the receptor modelling (section 3.6). 12 hour averages were generated based on the hourly values corresponding to the data collection period for the filters (from 10 am to 10 pm).

3.5 Receptor modelling using Positive Matrix Factorization (PMF)

Positive Matrix Factorization (PMF) (Paatero, 1993 and Paatero and Tapper, 1993, 1994) is a method for identification of sources for particulate matter and source contribution estimations. The PMF model use measured concentrations and uncertainty estimates to generate chemical profiles and time series associated with each profile for the filters collected. Chemical analysis of components and elements of measured PM samples are input to the model for identifying the emission sources. The method applied in this study is a 2 dimensional PMF model (PMF-2).

3.6 Emission sources classification

The receptor modelling resulted in factors, which from their chemical profiles where interpreted as representing certain source categories. Running the emission and dispersion models, the source categories were as far as possible divided into the same categories as in the receptor model (Table 1). The $PM_{2.5}$ sources in the receptor model were interpreted as long-range transport, wood burning or traffic related sources (Table 1). The classification of the emission sources for dispersion modelling includes additional anthropogenic sources. Sources included in this category were industrial sources, home heating using other fuels than wood burning, harbour and construction activities. In the PMF model, road dust and road salt are two different sources corresponding to re-suspension in the dispersion model.

Table 1: Classification of emission sources of $PM_{2.5}$ by the Positive Matrix Factorisation (PMF) model and emission sources used for dispersion modelling.

		PMF	Dispersion modelling
Wood burning		X	X
Long range transport		X	X
Exhaust particles- gasoline vehicles		X	X
Exhaust particles - diesel vehicles		X	X
Re-suspension	**Road salt**	X	X
	Road dust	X	
Other anthropogenic sources			X

3.7 Uncertainty estimation

A number of statistical methods can be considered when estimating the dispersion model uncertainties. Some of these methods are discussed in the cross cutting issue uncertainty report (Air4EU – M2) compiled in the Air4EU project. In this case study some of these methods are applied to evaluate the model performance and to estimate uncertainty of the dispersion model estimates. The analyses are performed both before and after updating of the emission inventory to reveal any improvements resulting from the updated emission estimate. In addition to standard statistical parameters such as temporal means and correlation coefficients, we have also calculated uncertainty indicators assessing model performance according to the EU directives based on recommendations provided in the Air4EU documents, (Air4EU – M2 and Air4EU D6.2, Part II). Both Relative Maximum Error (RME) and Relative Percentile Error (RPE) are calculated for daily values of PM. The RME is also applied to long term averages. In addition, Root Mean Square Error (RMSE) and Normalised Root Mean Square Error (NRMSE) are calculated to estimate the typical absolute and relative error of the model. Maps using NRMSE as an indicator for the relative error of the model are multiplied with the model concentration fields to give and absolute uncertainty field. A description of the statistical parameters calculated is given in Appendix C.

An extensive analysis of the receptor model uncertainty has not been presented in this case study but such an analysis is clearly required for establishing qualitative uncertainty to the independent assessment of the emission inventory. A limited uncertainty analysis has been carried out in Hagen et al., (2005).

4 Results

The overall aim of this study is to make an independent assessment of the emission inventory for Oslo. This has been done by comparing source contributions of $PM_{2.5}$ estimated by two independent methodologies, dispersion and receptor modelling. In addition, one of the aims is to use these results to see if the emission inventory, and corresponding PM estimates using the dispersion model, can be improved upon.

Dispersion modelling of hourly values of $PM_{2.5}$ was performed from 1 November 2003 until 1 May 2004. 12 hour average values corresponding to the same period as the filter samples were generated from the modelled hourly concentrations at the same site, Rv4. 40 filters (12 hour mean) collected from January until end of April 2004 provide the basis for the comparison between the two methodologies.

In section 4.2 the source apportionment estimates from the receptor model for $PM_{2.5}$ is presented. Corresponding source apportionment estimates from the dispersion model are presented in section 4.3. Comparison of the results from the two methodologies are analysed in sections 4.4. Based on these analyses the emission inventory is empirically updated for single source categories and $PM_{2.5}$ estimates are recalculated for Oslo. Monitor data from three independent stations, Kirkeveien, Løren and Aker Hospital, in addition to RV4, have been used to validate the impact of the updated inventory. Statistical analyses, based on daily means from 1 November 2003 to 1 May 2004, are presented in section 4.6.5. In addition, the spatial distribution of the average calculated $PM_{2.5}$ concentration fields and corresponding uncertainty estimates are presented in these sections. The measured $PM_{2.5}$ values (12 hour mean) for the 40 filters collected winter/spring 2004 are compared with the estimated 12 hour means from the dispersion model to identify the day to day agreements. These results are presented in sections 4.5.

4.1 Interpretation of $PM_{2.5}$ sources using the receptor model.

The filter samples used in the PMF model were analysed as described in section 3.2. The PMF model interpreted six emission sources for $PM_{2.5}$ and the chemical composition for the emission sources are presented in Table 2.

Table 2: Elements and compounds used for identification and interpretation of source profiles for $PM_{2.5}$ by the PMF model.

	Road Dust	Exhaust particles from diesel vehicles	Regional background (Long Range Transport)	Exhaust particles from gasoline vehicles	Road salt	Wood burning for domestic heating
SO_4	X	X	X	X	X	
NO_3	X	X	X		X	X
NH¤			X			
Cl	X		X		X	
Na			X		X	X
Ca	X	X	X	X		X
K	X	X	X	X	X	X
Al	X			X	X	
Fe	X	X		X	X	
Mn						
Ti					X	
Cu		X			X	X
Zn	X					
Sb						
Pb						
Sr						
Ni						
OC	X	X	X		X	X
EC	X	X	X	X		X
Levo					X	X
Baa-C						
aaa-C						

4.2 Source apportionment of $PM_{2.5}$ using receptor modelling

On the average the measured $PM_{2.5}$ concentration was 26.9 µg/m^3 for the 78 filters collected during the measurement campaign. The measured $PM_{2.5}$ average for 40 filters collected from January 2004 until end of April 2004 was 23.9 µg/m^3 (22.9 µg/m^3 in the PMF model results). The average source strengths, in percentages and concentrations, for the identified sources are given in Figure 2 and in Table 3, respectively.

The average source strengths indicate that traffic related sources are dominating with approximately 60 %. Regional background and wood burning contribute with approximately 20 % each.

The distribution shows that re-suspension (road dust and road salt) and gasoline vehicle exhaust are the main traffic sources (Table 3). Gasoline vehicles contribute approximately twice as much as diesel vehicles.

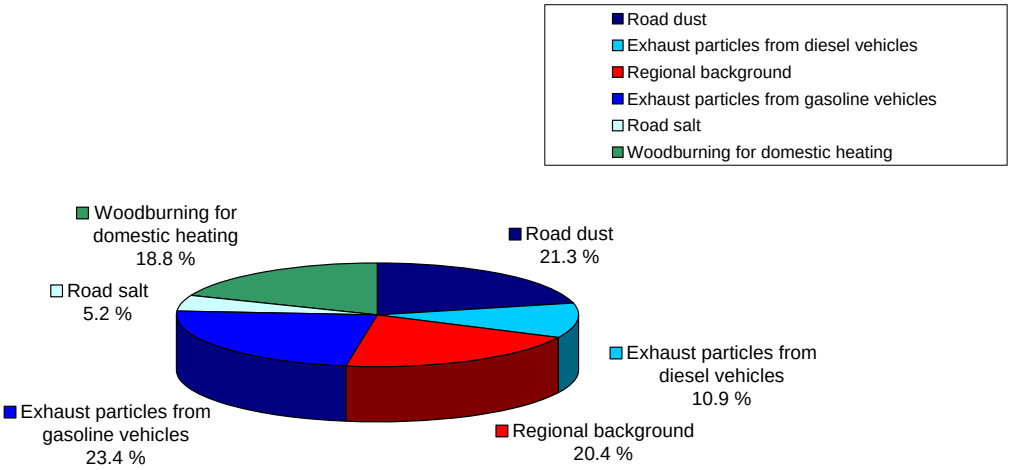

Figure 2: Average source contributions for $PM_{2.5}$ at Rv4 using PMF model, form January until end of April 2004, based on 12 hour means.

Table 3: Average concentrations and contributions to $PM_{2.5}$ for the sources interpreted from the PMF model, from January until end of April 2004.

	Total $PM_{2.5}$	**Road Dust**	**Road salt**	**Exhaust particles from diesel vehicles**	**Exhaust particles from gasoline vehicles**	**Wood burning for domestic heating**	**Regional background**
Concentration ($\mu g/m^3$)	22.9	4.9	1.2	2.5	5.4	4.3	4.7
Percentage (%)		21.3	5.2	10.9	23.4	18.8	20.4

4.3 Source apportionment of $PM_{2.5}$ using dispersion modelling

To estimate the source strength from the different emission sources, the dispersion model was run separately for each source category. 20.9 $\mu g/m^3$ was the average calculated $PM_{2.5}$ concentration corresponding to the measurement period for the filters in the PMF analysis.

The source apportionment using the dispersion model indicates that the most important $PM_{2.5}$ sources are wood burning (34%) and traffic related sources (33%), followed by regional background (32%) (Figure 3, Table 4). For traffic related sources, the vehicle exhaust is much more important than re-suspension.

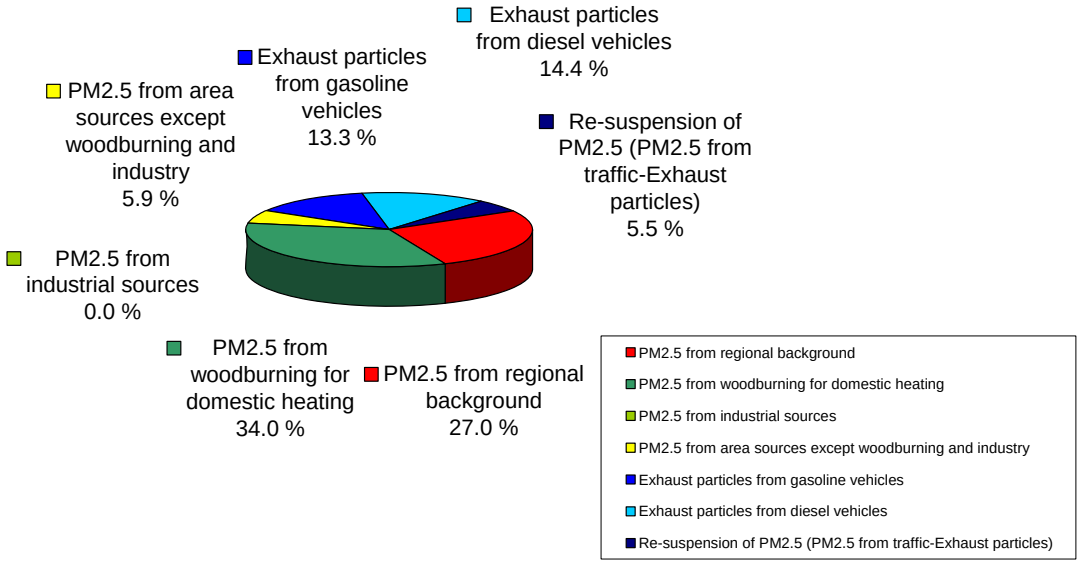

Figure 3: Average source contributions of $PM_{2.5}$ (based on 12 hour means) at Rv4 for selected days in winter/spring 2004 corresponding to PMF analysis.

Table 4: Average concentrations and source contributions to $PM_{2.5}$ estimated by the dispersion model, winter/spring 2004.

	Total $PM_{2.5}$	**Re-suspension**	**Exhaust particles from diesel vehicles**	**Exhaust particles from gasoline vehicles**	**Wood burning for domestic heating**	**Regional background**	**Area sources except wood burning and industry**	**Industrial sources**
Concentration ($\mu g/m^3$)	20.9	0.9	2.4	2.2	9.2	5.0	1.3	0.0
Percentage (%)		5.5	14.4	13.3	34.0	27.0	5.9	0.0

The 12 hour mean distributions for the individual source contributions, in concentrations, are shown in Figure 4. As expected, concentrations resulting from wood burning decrease towards the end of the heating season as the consumption are reduced. A time dependent pattern for the other sources, for instance for re-suspension, is not easy to find since this source varies with road surface conditions (dryness and salting). However, regional background episodes can easily be identified and the relative contribution of the regional background increases with time due to reduced contributions from wood burning.

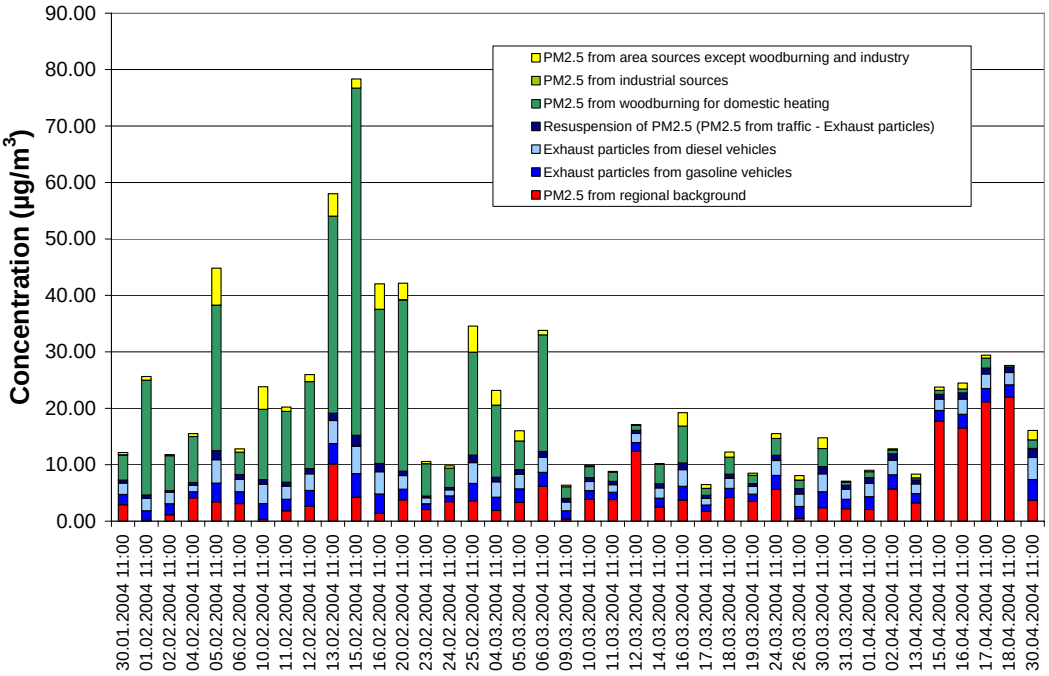

Figure 4: 12 h mean source distributions of $PM_{2.5}$ at Rv4 (µg/m^3) for selected days from January until end of April 2004 (corresponding to PMF analysis) calculated by the dispersion model.

4.4 Comparison of individual source strengths for $PM_{2.5}$ from receptor and dispersion modelling

The agreement between the dispersion model and the PMF model has been evaluated based on the average source distributions presented in section 4.2 and section 4.3. In addition, the 12 hour mean concentrations from the dispersion model have been compared with the 12 hour PMF model results for each source category.

The overall agreement between the dispersion model and the PMF model is very good. The average concentrations for the campaign period were 20.9 µg/m^3 and 22.9 µg/m^3, respectively. Even if the general agreement between the two methodologies is very good, there are differences for individual source categories.

The largest gaps are for wood burning from domestic heating and traffic related sources (vehicle exhaust and re-suspension) (Figure 5). The traffic related sources in the PMF model is approximately twice the contribution calculated by dispersion modelling for this source. The largest deviation is for re-suspension. For wood burning, on the other hand, the dispersion model estimate is approximately two times the PMF model estimate. The relative distribution between the source contributors will, consequently, not be the same for these two methodologies.

The regional background estimated by the receptor and dispersion models are on the average in good agreement (Table 3,Table 4, Figure 5). Even if large deviations occur on individual days (Figure 13) the overall correlation coefficient is 0.7.

For wood burning, as mentioned, the dispersion model estimates an average source strength approximately twice the receptor model (Table 3, Table 4, Figure 5). The correlation is high, 0.8. In general, the dispersion model estimates considerably higher concentrations on individual days during the heating season (Figure 14).

Traffic related re-suspension (road dust and road salt) of $PM_{2.5}$ is in poor agreement between these two methodologies. The PMF model estimates much larger source strength for re-suspension, approximately 6.7 times more than the dispersion model (Table 3, Table 4, Figure 5, Figure 17).

For gasoline vehicles the receptor model estimated more than twice the concentration estimated by the dispersion model. For diesel exhaust, there is very good agreement between the two methods (Figure 5, Figure 15, Figure 16). The PMF estimates for vehicle exhaust are encumbered with large uncertainties in regard to the differentiation of these sources (Hagen et al., 2005).

In addition to the sources interpreted by the receptor model, other anthropogenic sources such as industry, home heating using fuels such as heating oil, harbour and construction activities are defined sources in the emission inventory. These categories were defined as two additional sources in the emission inventory and dispersion calculations for $PM_{2.5}$: One called “Area sources except wood burning and industry” and one called “Industrial sources”. The latter is negligible in the results. The average contribution for “Area sources except wood burning and industry” source is 1.3 µg/m^3 (Figure 5).

The PMF model does not include any other anthropogenic combustion sources than traffic. It is assumed that these anthropogenic sources are included in the traffic estimates from the PMF model. To quantify this, contributions from exhaust vehicle emissions in the PMF model are compared with the contributions estimates for vehicle exhaust emission and area distributed combustion sources (except wood burning) from the dispersion model (source 3, 4, 5 and 6 in Figure 5). The average deviation is approximately 2 µg/m^3, where the PMF model estimates the highest concentration. This deviation is 9 % of the total average concentration.

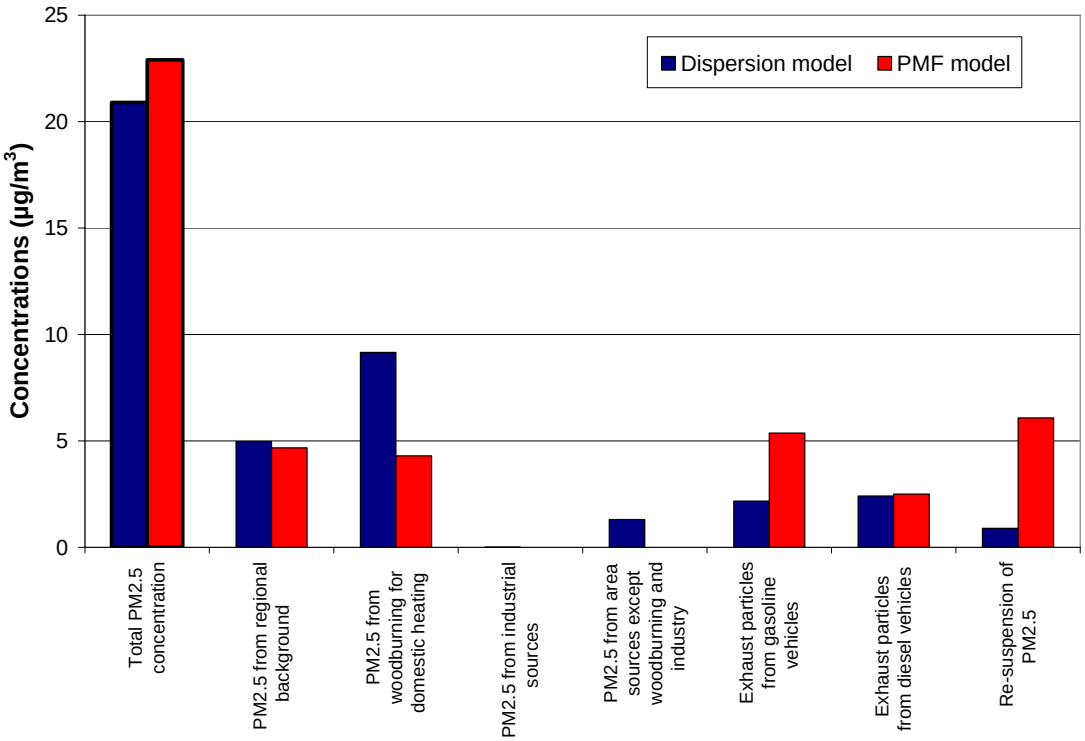

Figure 5: Estimated average concentrations of $PM_{2.5}$ ($\mu g/m^3$) from different source categories using dispersion (blue) and receptor models (red) at Rv4 winter/spring 2004.

4.5 Comparison of measured and estimated $PM_{2.5}$

The estimated $PM_{2.5}$ concentrations (12 hour averages) from the dispersion model are compared with the measured concentrations (12 hour averages) using KFG at Rv4 (Figure 6).

In general there is a very good agreement between the measurements and the dispersion model results. On the average the dispersion model underestimates the concentration with approximately 3 $\mu g/m^3$ or 13% (Table 5). There is some overestimation on single days in the winter months and generally some underestimation during the spring (Figure 6).

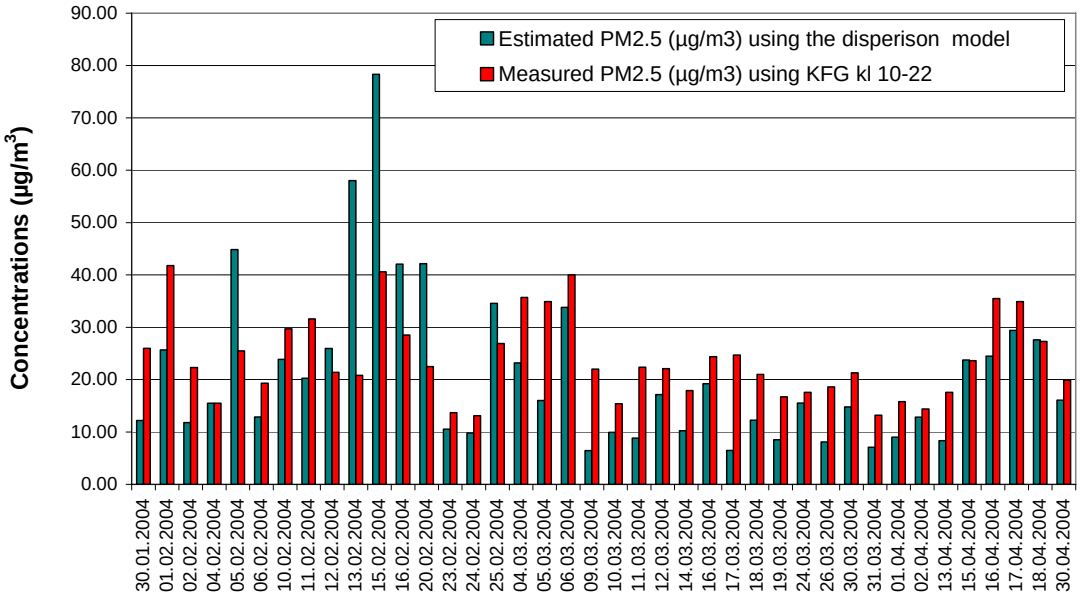

Figure 6: Calculated and measured (KFG) 12 hour averages (10-22) of $PM_{2.5}$ ($\mu g/m^3$) winter/spring 2004 (corresponding to days with filter analysis).

Table 5: Comparison between $PM_{2.5}$ estimates (12 hour means) and measurement (KFG) at Rv4, from January to end of April 2004 (corresponding to days with filter analysis).

	Measurements (KFG) $PM_{2,5}$ (10-22) ($\mu g/m^3$)	**Dispersion model $PM_{2.5}$ from all sources ($\mu g/m^3$)**
Average	23.9	20.9
Max	41.8	78.3
Min	13.1	6.4
Std. deviation	7.9	15.2
Corrcoef		0.55
RMSE		12.95
NRMSE		0.62

4.6 Assessment and updated estimates of $PM_{2.5}$

4.6.1 Wood burning

The analysis performed showed a relative large deviation, approximately 2 times, between the estimated source strength of $PM_{2.5}$ from wood burning using the dispersion model and PMF model (section 4.4, Figure 5).

The yearly emission/consumption data in the model calculations are not adjusted according to causality influencing the consumption of wood burning. One example is the temperature difference between the data collection year (2002) and the calculation year (2004). This might lead to an overall overestimation or underestimation in the emissions depending on the difference in consumption between these two years. In addition, the emission estimate does not take into account the difference between the daily temperature in the calculation year and for the 30 year temperature average used here as basis for the daily time variation. Based on the assumption that the consumption is temperature dependent, this might also lead to an underestimation or overestimation in the daily emissions.

In a previous study (Larssen et al, 2006), estimated $PM_{2.5}$ concentrations from wood burning using dispersion modelling were compared with measurements of a tracer for wood burning (levoglucosan). One of the aims in that project was to identify improvements of the calculated concentrations from wood burning taken into account real temperature variations in the model calculation year. The study demonstrated an overall improvement in the estimated source strength of wood burning if the emissions were adjusted according to the actual yearly average and daily average temperature variations for the calculation year.

Another uncertainty in the model calculation is the initial spatial distribution of emissions in the dispersion model. One important issue is for example the vertical grid size. To evaluate this effect, we performed a test where emission from wood burning was emitted in the lowest model layer (i.e. 14 m). On the average, the concentration increased with approximately 40 % (from 6 µg/m^3 to 10 µg/m^3) compared to estimated concentrations where wood burning was emitted in the lowest three model layers i.e. 72 m (normally used for Norwegian cities). Thus, the concentration level is highly dependent on how the emissions are put into the model.

The analysis performed in this study indicates that the dispersion model overestimates concentrations resulting from wood burning. The discussion above indicates that this may be attributed to inaccuracies in emission rates or due to inaccuracies in model formulation, or both. In regard to updating the emission inventory it is assumed that the difference between receptor and dispersion modelling is due to the emissions inventory only. A new emission estimate for wood burning for domestic heating has therefore been generated based on the PMF model results. The emissions were reduced empirically by a factor of two.

4.6.2 Regional background

Measurements of $PM_{2.5}$ from a regional background station, Birkenes, was used as regional background in the dispersion model (section 3.1). On the average the agreement between the receptor model and the dispersion model is good and the day to day variation is also fairly well described. However, a more detailed analysis should be performed on the representativity of this station. Agreement and deviations day by day should be compared with the synoptic situation to identify the general atmospheric circulation patterns influence on the representativity of this station as a regional background station for Oslo.

4.6.3 Exhaust particles and other anthropogenic sources

The vehicle exhaust emission is calculated based on annual daily traffic (ADT), vehicle distribution on each road link and emission factors for various emission vehicle classes (section 3.3). The results indicate a very good agreement on the average between the receptor model and dispersion model for diesel vehicles. For gasoline vehicles, however, the dispersion model underestimates the concentrations compared to the receptor model. The uncertainties in the estimated concentrations from diesel and gasoline vehicles are large in the PMF results (Hagen et al, 2005).

As previously mentioned, the emission inventory also includes two additional sources not identified by the PMF model, other anthropogenic sources and industrial activities. On the average, 1.3 µg/m^3 or

6 % of the total concentration originate from these two sources. It is expected that these source categories are included in the traffic related sources in the receptor model results.

4.6.4 Re-suspension

As presented in section 3.3, re-suspension is dependent on the percentage of studded tyres, road surface condition and level of the exhaust particles. This is not a very good description of re-suspension of $PM_{2.5}$ since the re-suspension will decrease with implementation of new vehicle technology. The estimated re-suspension (dust and road salt) of $PM_{2.5}$ from the receptor model is approximately 6.7 times higher than the dispersion model. In order to improve the results from the dispersion model, new estimates of re-suspension emission has been made my multiplying the emission from re-suspension of $PM_{2.5}$ by 6.7.

4.6.5 Recalculated estimates of $PM_{2.5}$

Based on the comparison of the PMF and dispersion modelling results at Rv4 and the discussion above, it was decided to update empirically the emission inventory for two sources, wood burning and re-suspension of $PM_{2.5}$. Updated estimates of $PM_{2.5}$ concentrations were calculated for Kirkeveien, Løren, Aker hospital and Rv4 after recalculation of the source strength for these two sources. The same scaling was done on daily (24 hours) values from 1 November 2003 to 1 May 2004. The emission from wood burning was reduced by 2.0 and the emission for traffic induced re-suspension was increased by 6.7. Measurements from these independent stations where used to validate the impact of the updated emission inventory. Statistics, based on daily means, were generated for all the four stations. Figures showing the estimated daily values before and after the updating of the emission inventory for $PM_{2.5}$ and corresponding scatter plots for all the stations, are presented in Appendix A.

In general, the statistics are improved at all sites (Appendix D, Table 6, Table 7, Table 8, Table 9). For all the stations evaluated the correlation coefficient increased after updating the emissions. The average estimated concentration increased or decreased. The concentrations are dependent on which sources are dominating at the specific location. For instance at Løren, which is dominated by traffic emissions, the average value increases. At Kirkeveien, on the other hand, which is a location dominated both by wood burning for domestic heating and traffic, the average value decreases. At the urban background station, Aker Hospital, the average value also decreased. The maximum deviation between average observed and modelled $PM_{2.5}$ concentrations at all stations was, however, less than 15%.

The statistical methods for estimating model uncertainties according to EU directives (Air4EU – M2) have been applied both before and after updating of the emission inventory to reveal if there were improvements by updating emission estimates (Appendix D, Table 6, Table 7, Table 8, Table 9).

The Relative Maximum Error (RME) and the Relative Percentile Error (RPE) for the 36^{th} (EU directives) and the 8^{th} (Norwegian guidelines) highest daily values are predicted. The results show in general good agreement between the measured and estimated percentile values. The average Relative Maximum Error ranges from 20-28% for the 180 daily values. The error for the 36^{th} highest and the 8^{th} highest daily values ranges from 9 to 64 %. The Relative Maximum Error is as large as 1.3 before correction of the emission estimates. After update of the emissions of wood burning and re-suspension of $PM_{2.5}$ the percentile estimates are also improved. The largest RME and the $RPE_{36,8}$ is 0.67 and 0.35, respectively.

Figure 7 presents the average concentration maps in $1km^2$ grid cells from 1 November 2003 to 1 May 2004 (a) before and (b) after updating of the emission inventory. The area distributed concentrations generally decrease due to reduction in the spatial distribution of wood burning for domestic heating. On the other hand, close to the road link network, the concentrations increases due to the increase of the source strength of traffic induced re-suspension of $PM_{2.5}$. Depending on the most important local

sources, traffic or wood burning for domestic heating, the calculated concentrations either increases or decrease.

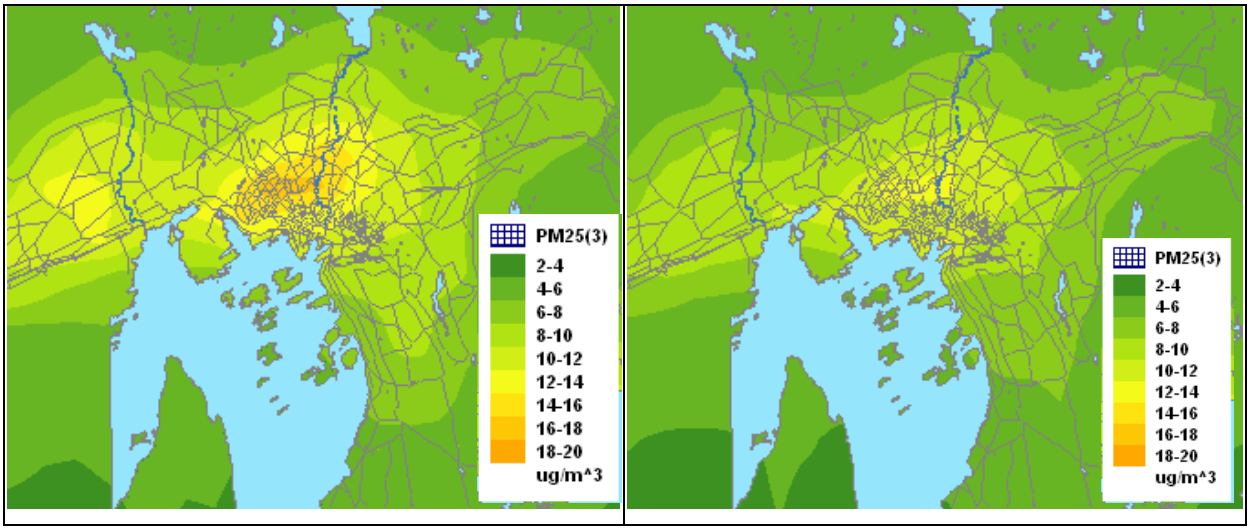

Figure 7: a) Average $PM_{2.5}$ from 1 November 2003 to 1 May 2004. b) New estimate of average $PM_{2.5}$ from 1 November 2003 to 1 May 2004 after updating of the emission inventory.

Based on the estimated Normalise Root Mean Square Error (NRMSE), which is representative for the typical relative error, the maps showing the absolute uncertainties generated (a) before and (b) after the updated $PM_{2.5}$ concentration estimates are given in Figure 8. The NRMSE is calculated based on the average concentration for the calculation period at the four stations, Kirkeveien, Løren, Rv4 and Aker hospital. The NRMSE was 7.5% before the update of the emission inventory and 9.6% after the update of the inventory, a slight increase in the relative uncertainty. The maximum error is observed in the area of highest concentration, that being the city center of Oslo.

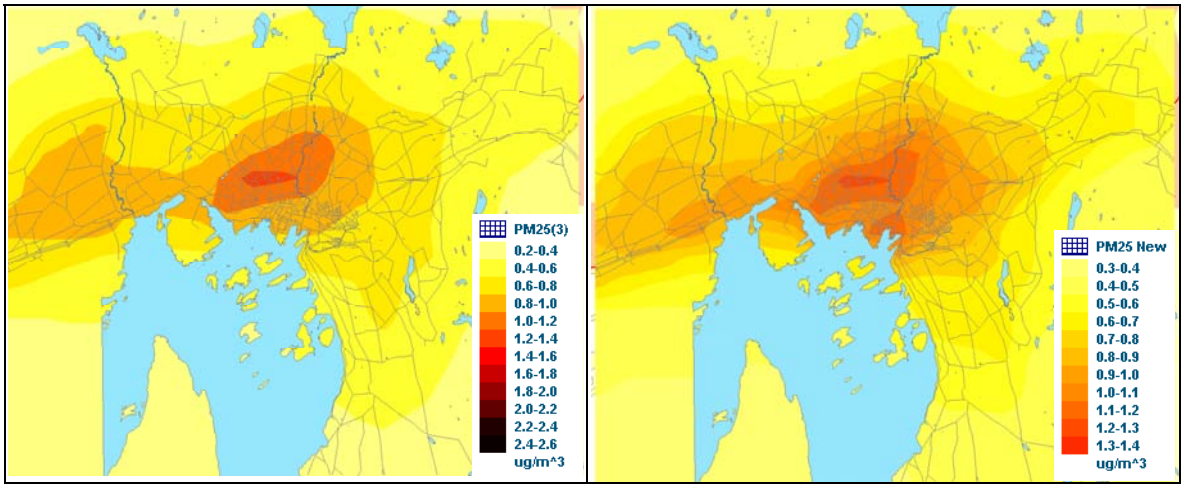

Figure 8: Estimate uncertainties ($\mu g/m^3$) of the $PM_{2.5}$ from 1 November 2003 to 1 May 2004 before (a) and after (b) the emission inventory was updated.

4.7 Comparison of the estimated source contribution of $PM_{2.5}$ at four stations

Comparison of the source contributions at Kirkeveien. Løren, Rv4 and Aker hospital before and after the update of the emission inventory have been performed (Figure 9, Figure 10, Figure 11, Figure 12). This has been done to identify the relative source contributions as well as the total concentration estimate of $PM_{2.5}$ at different sites.

At Kirkeveien both wood burning and traffic are important emission sources. The reduction in emission from wood burning is greater then the increase in traffic induced re-suspension. The average $PM_{2.5}$ concentration lowers with approximately 2 µg/m^3 after the update of the inventory. Løren and Rv4 are both road side stations. Løren has highest traffic impact since the traffic density is approximately two times the annual daily traffic of Rv4. However, the impact at Løren is not much higher than at Rv4. This is because the impact of the traffic emissions (exhaust and re-suspension) is strongly dependent on the distance from road side. RV4 is located approximately 5 m and Løren 15 m from road in the model. At both sites, the reduction in impact from wood burning is compensated by the increase in traffic induced re-suspension. The average concentration is more or less the same after the updating of the emissions. Aker hospital is an urban background station and road traffic has less impact on this site. The average $PM_{2.5}$ concentration is therefore reduced after the scaling of wood burning and re-suspension.

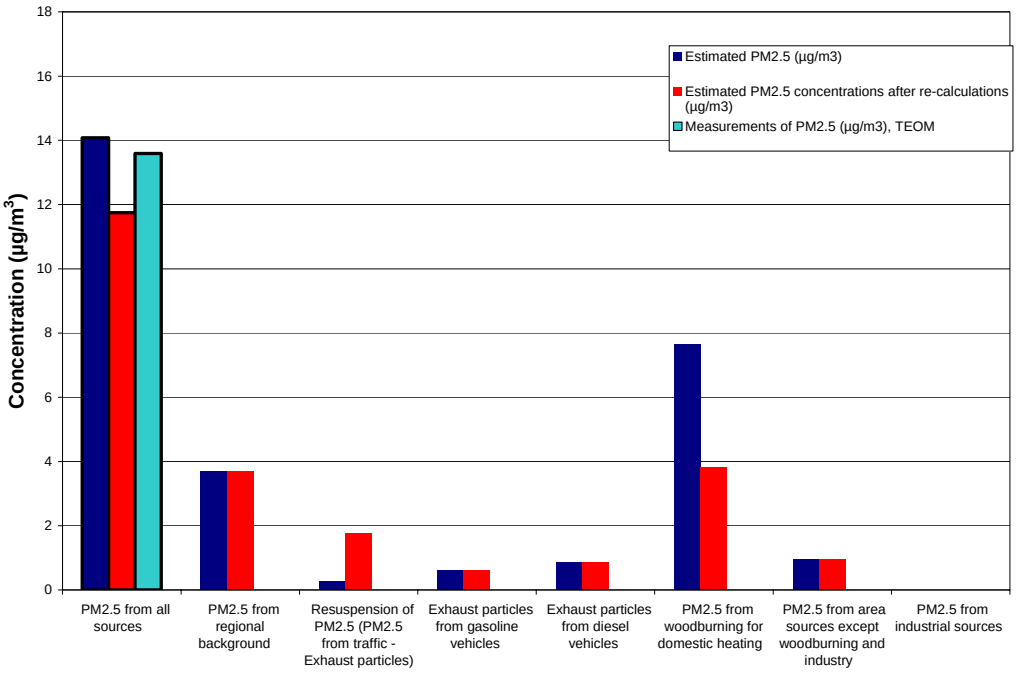

Figure 9: Average estimated and measured $PM_{2.5}$ concentrations and corresponding source contributions before and after update of the emission inventory from 1 November 2003 to 1 May 2004 at Kirkeveien.

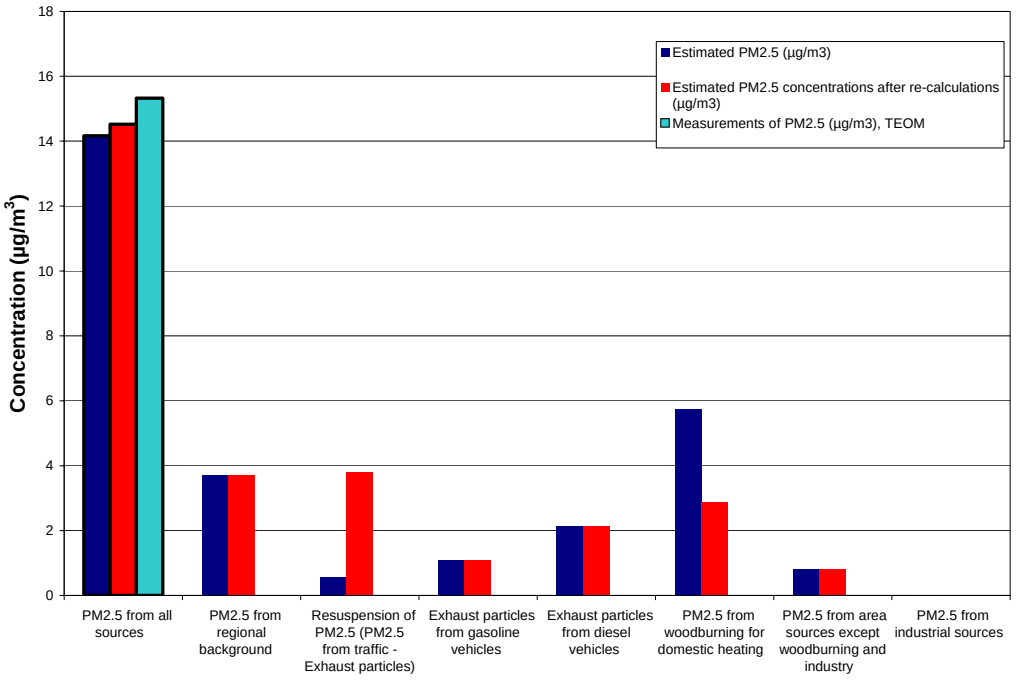

Figure 10: Average estimated and measured $PM_{2.5}$ concentrations and corresponding source contributions before and after update of the emission inventory from 1 November 2003 to 1 May 2004 at Løren.

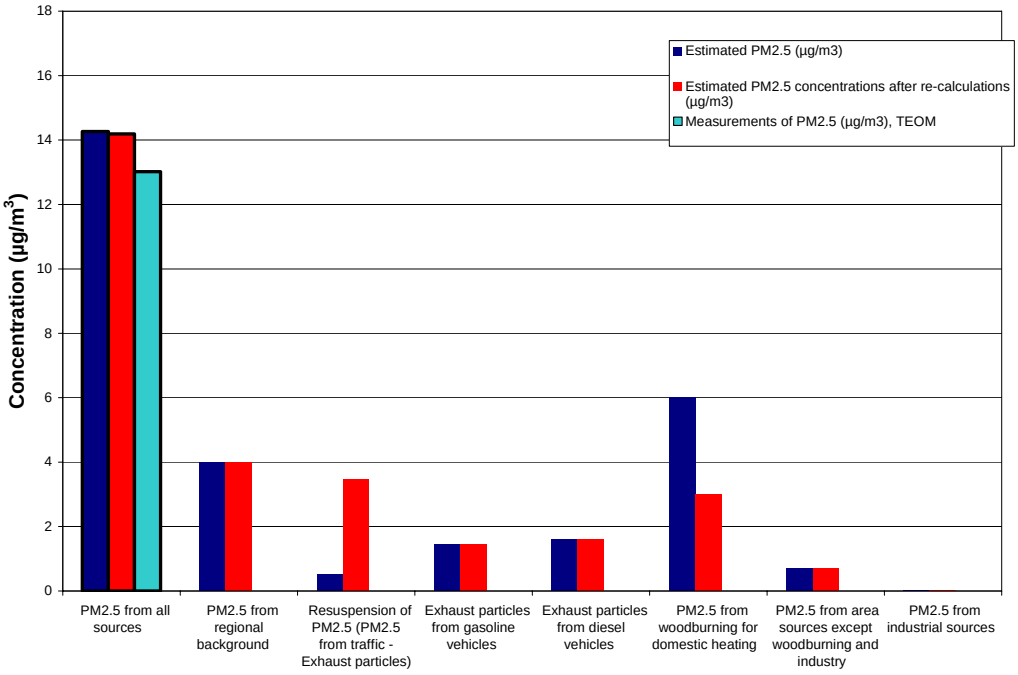

Figure 11: Average estimated and measured $PM_{2.5}$ concentrations and corresponding source contributions before and after update of the emission inventory from 1 November 2003 to 1 May 2004 at Rv4.

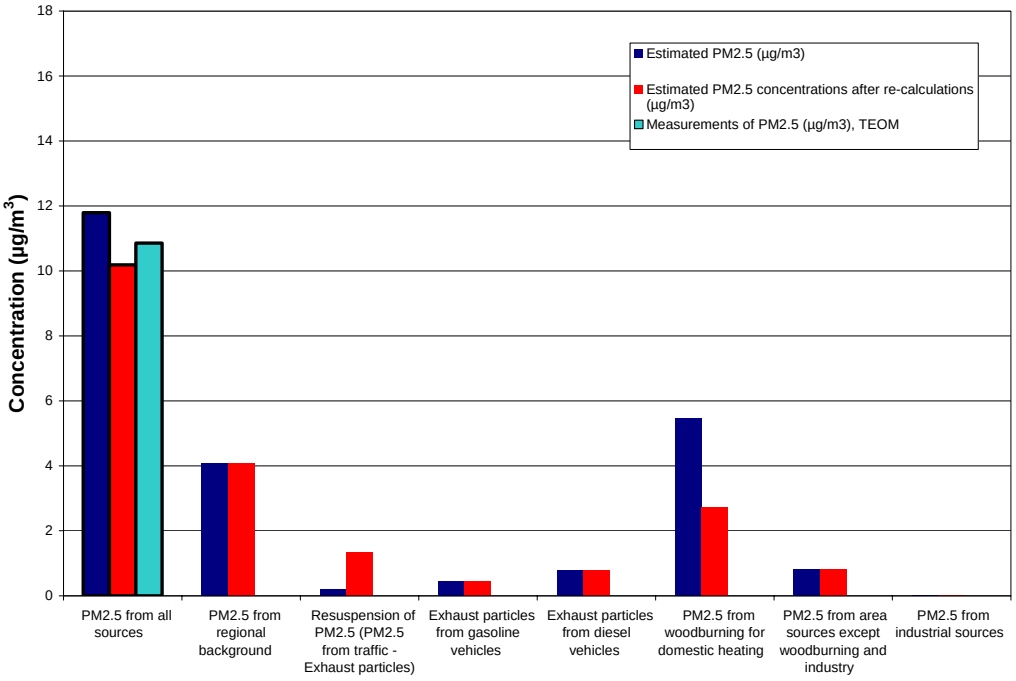

Figure 12: Average estimated and measured $PM_{2.5}$ concentrations and corresponding source contributions before and after update of the emission inventory from 1 November 2003 to 1 May 2004 Aker Hospital.

5 Conclusion and discussion

A case study for an independent assessment of emission inventories has been performed within the scope of the EU-funded project Air4EU. In this case study, the quality of an emission inventory for particulates ($PM_{2.5}$) has been assessed. This assessment has been performed by comparing source apportionment estimates from a dispersion model using an emission inventory with source apportionment estimates from a receptor model (Positive Matrix Factorisation). In addition, one of the aims with this case study was to use the outcome of the comparison of the two methodologies to improve the emission inventory and the associated PM estimates. Based on the analysis, the emission inventory was updated for individual source categories and $PM_{2.5}$ concentrations were re-calculated to reveal improvements in the results.

5.1 Assessment of the case study

The comparison between the two methodologies identified gaps as well as weaknesses in the emission inventory and in the receptor model results.

For $PM_{2.5}$ the receptor model interpreted four different traffic related sources, regional background and wood burning for domestic heating. The emission inventory and the dispersion model calculations where as far as possible classified using similar categories. The most dominating source in the PMF model results was identified to be traffic related. For the dispersion model wood burning for domestic heating was dominant.

For $PM_{2.5}$ the largest deviation was observed for two sources, wood burning for domestic heating and traffic induced re-suspension. Traffic induced re-suspension was underestimated compared to the receptor model by a factor 7 and wood burning for domestic heating was overestimated approximately

by a factor of 2. The average estimated concentration, for the period where the 40 filters were collected, from wood burning for domestic heating was approximately 9.1 µg/m^3 from the dispersion model and 4.3µg/m^3 from the PMF model. For the traffic induced re-suspension the average concentrations were 0.89µg/m^3 and 6µg/m^3, respectively. For the other sources, regional background, vehicle exhaust together with other anthropogenic combustion sources, agreed fairly well.

Based on the analysis and uncertainties in the emission estimates there are indications that improvements in the emission inventory might lead to improved concentration estimates of $PM_{2.5}$. Uncertainties in the total consumption of wood burning for domestic heating due to temperature differences between the inventory year and the calculation year or other causality influencing the consumption data, for example electricity rates, are examples of uncertainties in the emission inventory. Another uncertainty is how the dispersion model apply these emissions e.g. how the emissions are put into the model.

The empirical relation used in the emissions model for traffic induced re-suspension of $PM_{2.5}$ has known weaknesses, which support the observed deviation between the two methodologies. In addition, only a limited number of sources related to traffic, regional background and wood burning for domestic heating are interpreted by the receptor model. In other words, sources not related to these categories are incorrectly included in these source contribution estimates.

5.2 Improvements in assessment derived from the case study

Based on the analysis and the interpretation of the results the emission inventories for$PM_{2.5}$ were updated to identify if this could lead to improvements of the concentration estimates of $PM_{2.5}$ at other stations in Oslo.

The largest gaps for $PM_{2.5}$ are identified for two sources, wood burning and re-suspension. Based on the average deviation between the PMF model and the dispersion model, we rescaled the emission inventory for individual source categories and calculated new $PM_{2.5}$ estimates for Oslo using the dispersion model. In regard to updating the emission inventory it is assumed that the difference between receptor and dispersion modelling is due to the emissions inventory only.

The new estimated $PM_{2.5}$ concentrations were compared with measurements at three other independent stations to evaluate the improvements of the updated inventory. The updated estimate of $PM_{2.5}$ concentrations were calculated for Kirkeveien, Løren, Aker hospital and Rv4 after recalculation of the source strength for the individual sources. The same scaling was done on daily (24 hours) values from 1 November 2003 to 1 May 2004 based on the analysis performed at Rv4. New statistics were performed for all the four stations.

For $PM_{2.5}$ the standard statistic parameters for model evaluation and statistic parameters for evaluation of model calculations according to EU directives are in general improved at all sites. For all the stations the correlation coefficient (based on daily means) increases after updating the emissions. The average value increases or decreases dependent on which sources dominates at the specific locations. The new estimated average concentration map showed a reduction in the spatial concentration values of $PM_{2.5}$. This is probably due to the reduction of the source strength of wood burning for domestic heating, which is a spatial distributed source. The source strength for traffic induced re-suspension is however a local hot spot source and will therefore have less influence on the spatial distributed concentrations.

Even if the measured and modelled concentrations are in good agreement, the analysis has shown the importance of a well described source contribution. .

$PM_{2.5}$ is an example on how updated emission estimates for single sources may have significant impact on the spatial concentration estimates and the associated exposure estimates and abatement measures.

5.3 Recommendations resulting from the case study

Source apportionment of PM using receptor modelling provides a good basis for independent assessment of emission inventories and is a recommended methodology for advanced assessment. Even though both dispersion and receptor modelling have weaknesses, comparisons of source apportionment estimates reveal both deviations and conformities and are a very good basis for a detailed analysis of the results. However, it is important that the tracer components and source profiles from the receptor modelling are well defined when these data are used to improve the emission inventory. If not, this might result in incorrect updates of the individual sources both in time and space. If such a methodology is applied to update emission inventories then independent assessment at other stations is required to assure that the update does indeed lead to improved estimates of concentrations.

One of the more practical recommendations coming from this study is the need to collect large enough samples or parallel sampling to reduce uncertainty in the chemical analysis. Since a large number of chemical compounds are analysed, and not all in the same instrument, the total mass available for analysis may lead to significant reductions in accuracy of the methods. A pre-study is recommended to assess the requirements for the monitoring that will provide sufficient accuracy for the assessment.

When carrying out such a study it is also important to critically assess uncertainties coupled to the dispersion model itself and not simply assume that the model is correct. As pointed out model calculated concentrations can be sensitive to model formulation, in this case to the vertical distribution of domestic heating emissions which is an uncertain aspect of the modelling. Other aspects, such as turbulence parameterisations, may also lead to bias in model calculated concentrations that can not be separated from emission rates without comprehensive vertical profile measurements.

It is also recommended that, to give a clearer picture of the urban scale concentrations in regard to model validation, a suitable number of urban background stations be deployed in the urban region. For the case of Oslo there were 4 traffic stations and 1 urban background station measuring $PM_{2.5}$ during the study period. Since the most important source to $PM_{2.5}$ is domestic wood burning, and thus a spatially distributed source, it would make more sense to measure $PM_{2.5}$ at a larger number of urban background stations than were available for this study.

5.4 Suitability for implementations in other cities

Receptor modelling and source apportionment studies of PM is suitable for application in any city where more than one source of PM is expected to contribute significantly to concentrations and where there is uncertainty in the strength of these sources. Before carrying out any such study though suitable chemical species must be selected for analysis and a suitable site(s) must be found that will be representative of the sources under consideration.

Such a study also requires a high level of expertise, in the chemical analysis, in the dispersion modelling, in the receptor modelling, and in the analysis. Cities wishing to undertake such activities must have such expertise available.

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Appendix A

Comparison of source strengths of $PM_{2.5}$ estimated by PMF and dispersion model

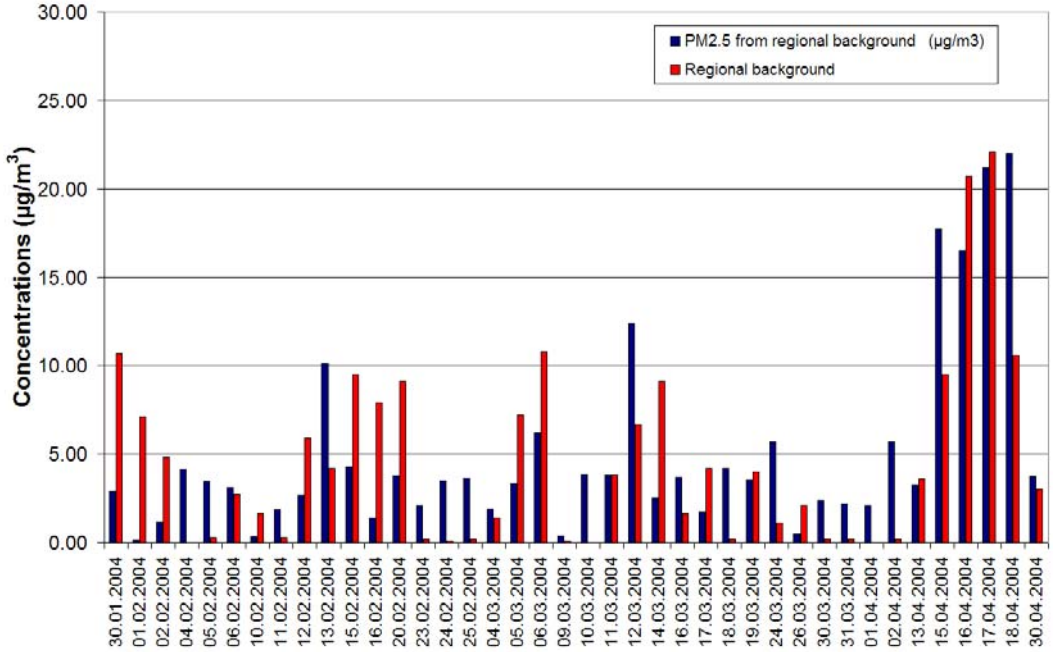

Figure 13: Estimated 12 hour mean concentrations of $PM_{2.5}$ from regional background (µg/m^3) using dispersion (blue) and receptor model (red).

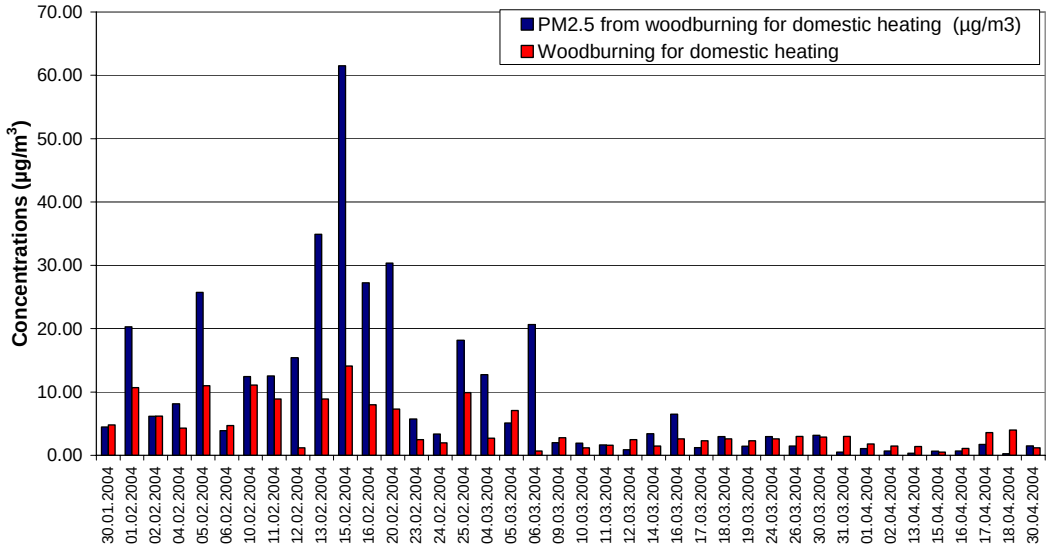

Figure 14: Estimated 12 hour mean concentrations of $PM_{2.5}$ from wood burning (µg/m^3) using dispersion (blue) and receptor model (red).

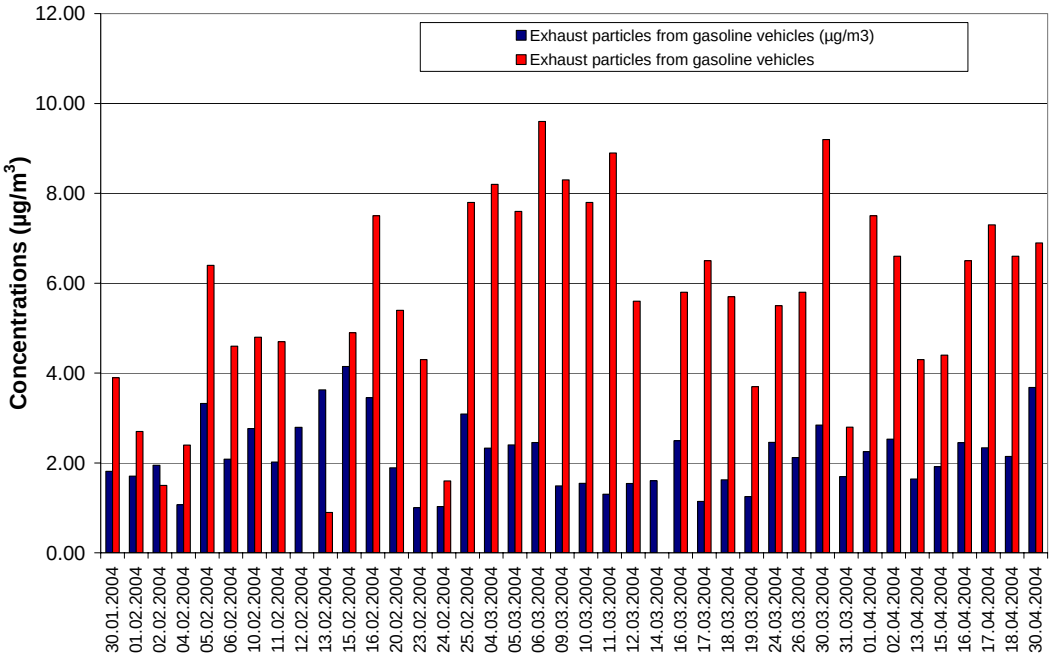

Figure 15: Estimated 12 hour mean concentrations of $PM_{2.5}$ from exhaust particles ($\mu g/m^3$) from gasoline vehicles using dispersion (blue) and receptor models (red).

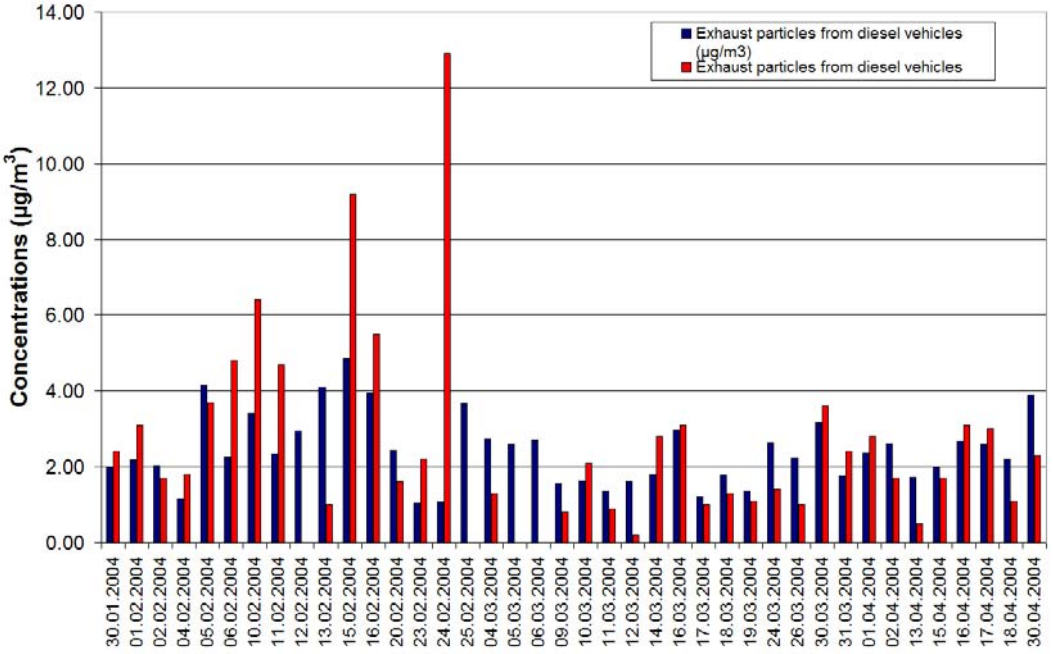

Figure 16: Estimated 12 hour mean concentrations of $PM_{2.5}$ from exhaust particles from diesel vehicles ($\mu g/m^3$) using dispersion (blue) and receptor models (red).

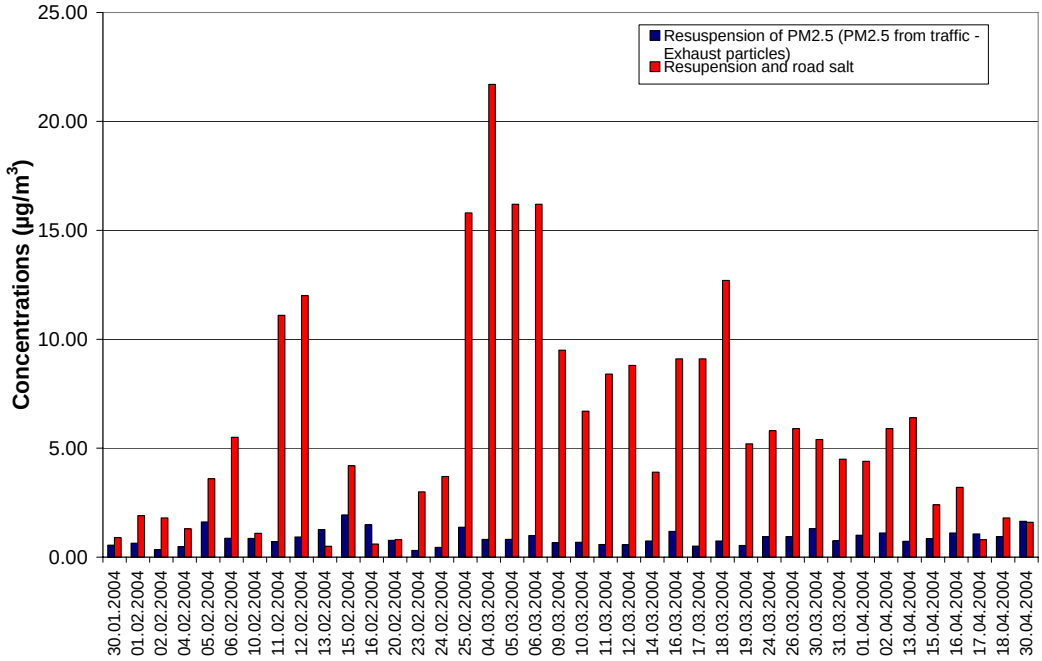

Figure 17: Estimated 12 hour mean concentrations of $PM_{2.5}$ from re-suspension (µg/m^3) using dispersion (blue) and receptor models (red).

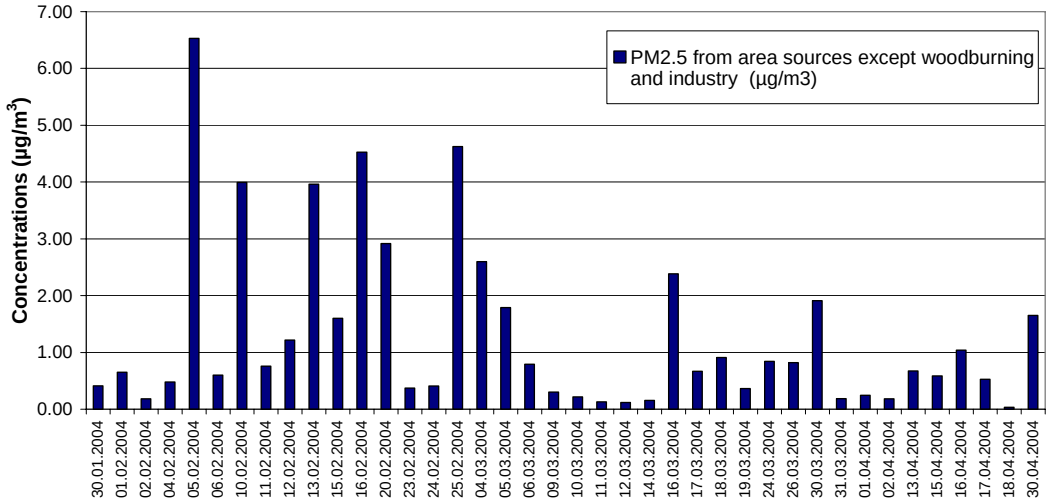

Figure 18: Estimated 12 hour mean concentrations of $PM_{2.5}$ from area source except wood burning and industry (µg/m^3) using dispersion (blue).

Appendix B

Updated estimates of $PM_{2.5}$

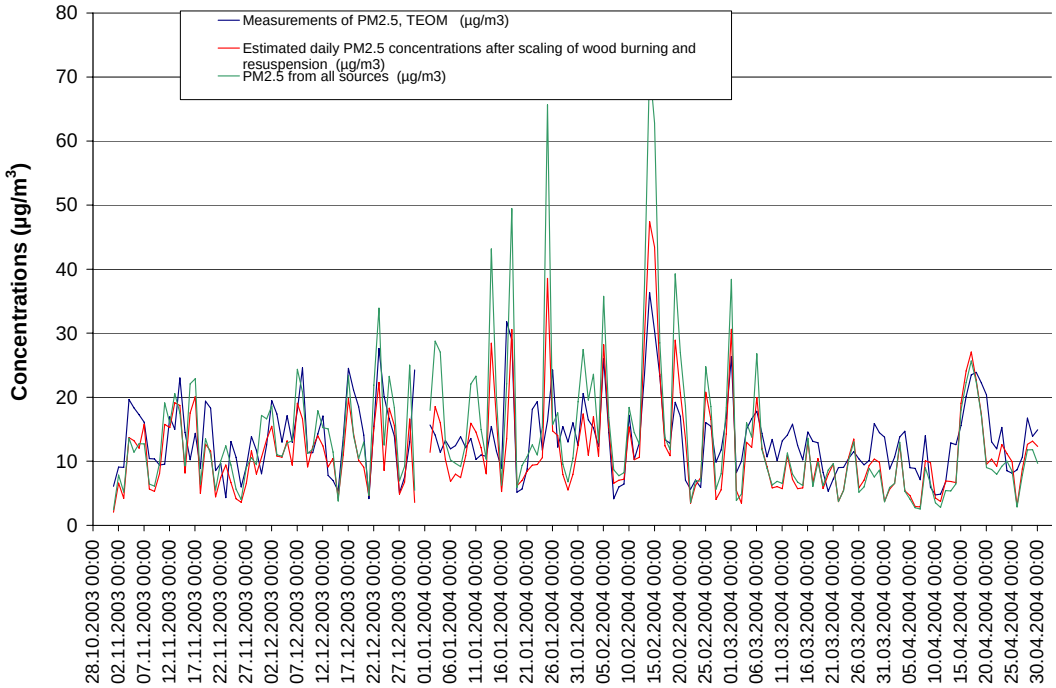

Figure 19: Estimated and measured daily $PM_{2.5}$ at Kirkeveien 01.11.2003-01.05.2004, before and after scaling of wood burning and re-suspension.

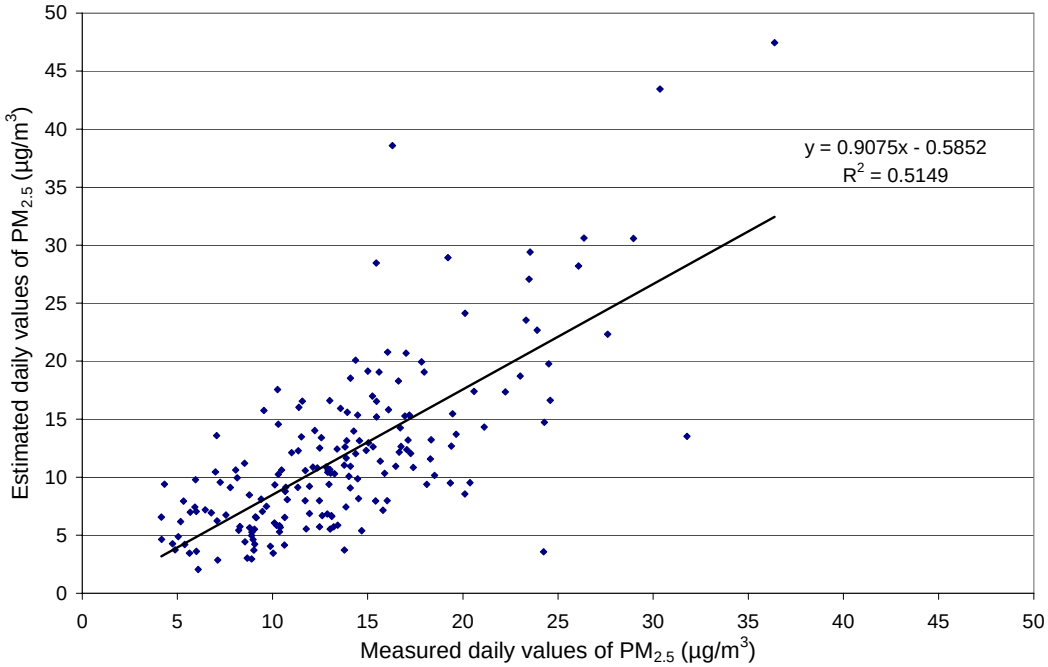

Figure 20: Estimated and measured daily $PM_{2.5}$ at Kirkeveien 01.11.2003-01.05.2004 after scaling of wood burning and re-suspension.

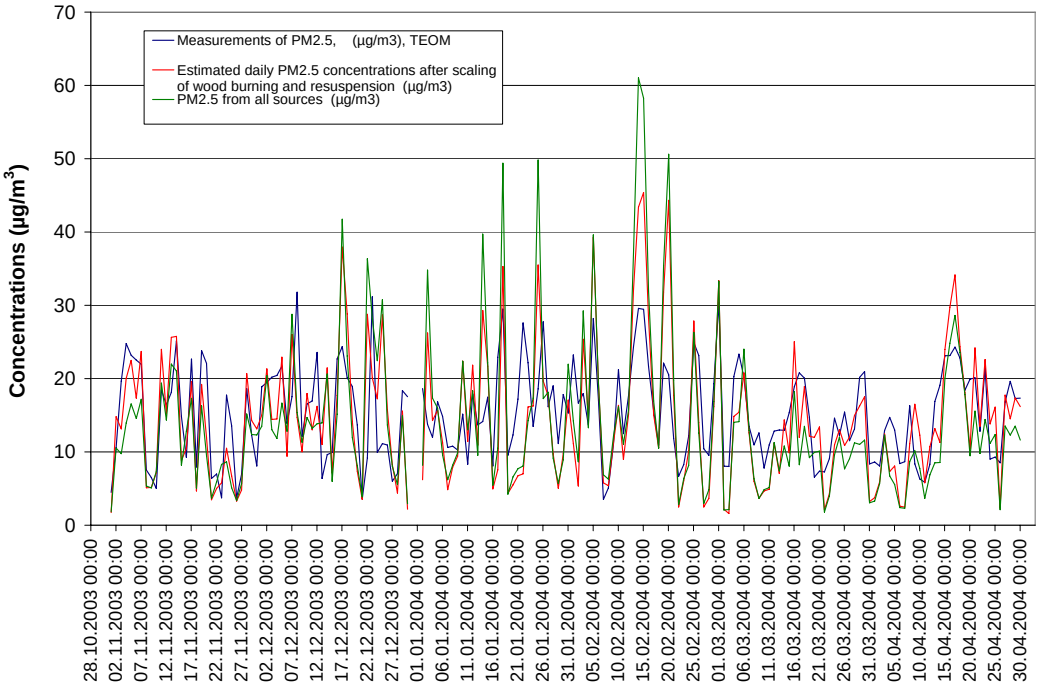

Figure 21: Estimated and measured daily $PM_{2.5}$ at Løren 01.11.2003-01.05.2004, before and after scaling of wood burning and re-suspension.

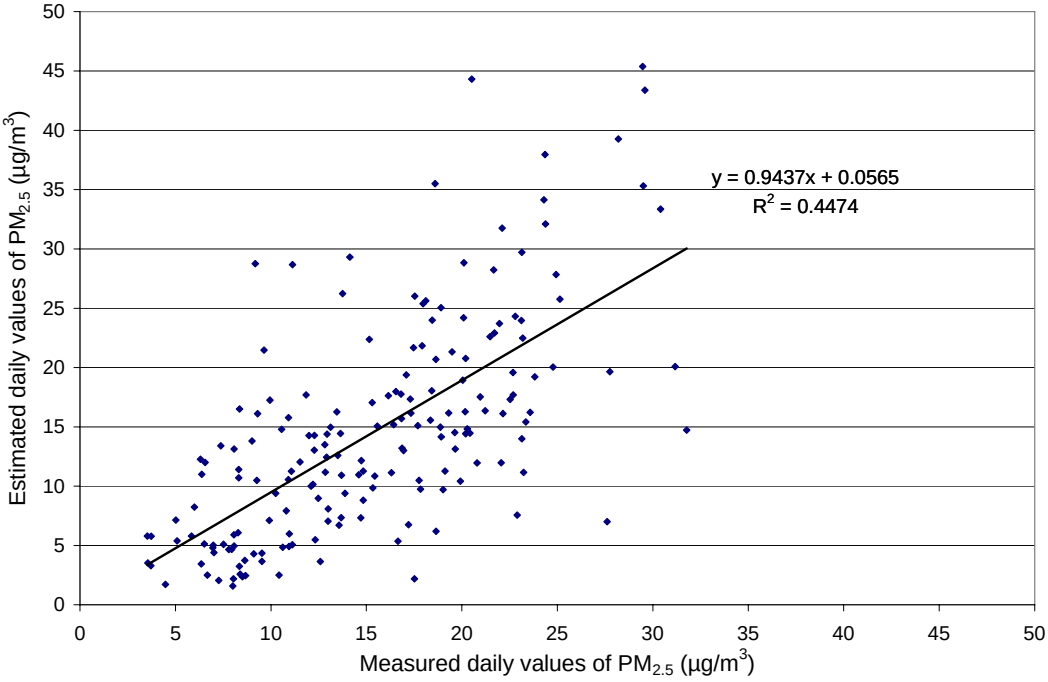

Figure 22: Estimated and measured daily $PM_{2.5}$ at Løren 01.11.2003-01.05.2004 after scaling of wood burning and re-suspension.

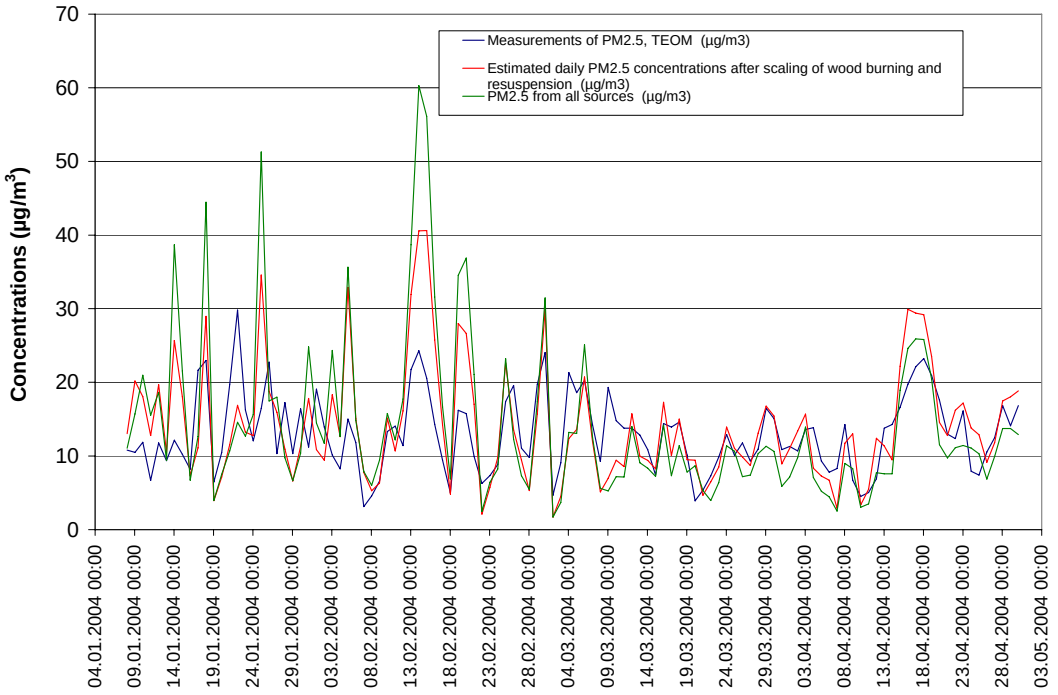

Figure 23: Estimated and measured daily $PM_{2.5}$ at Rv4 08.01.2004-01.05.2004, before and after scaling of wood burning and re-suspension.

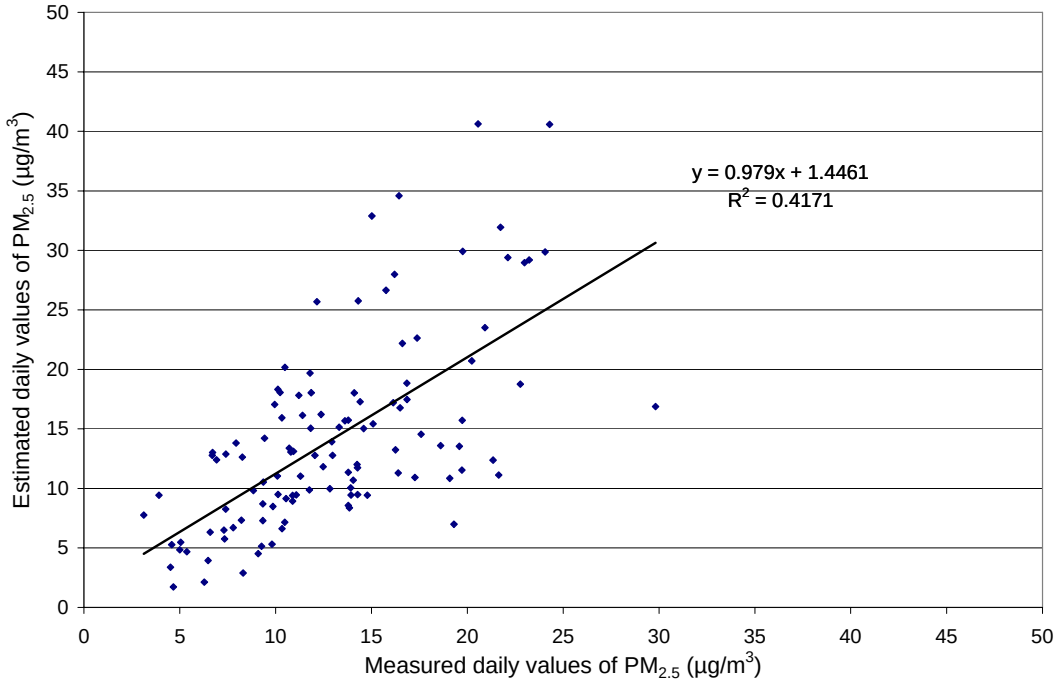

Figure 24: Estimated and measured daily $PM_{2.5}$ at Rv4 08.01.2004-01.05.2004 after scaling of wood burning and re-suspension.

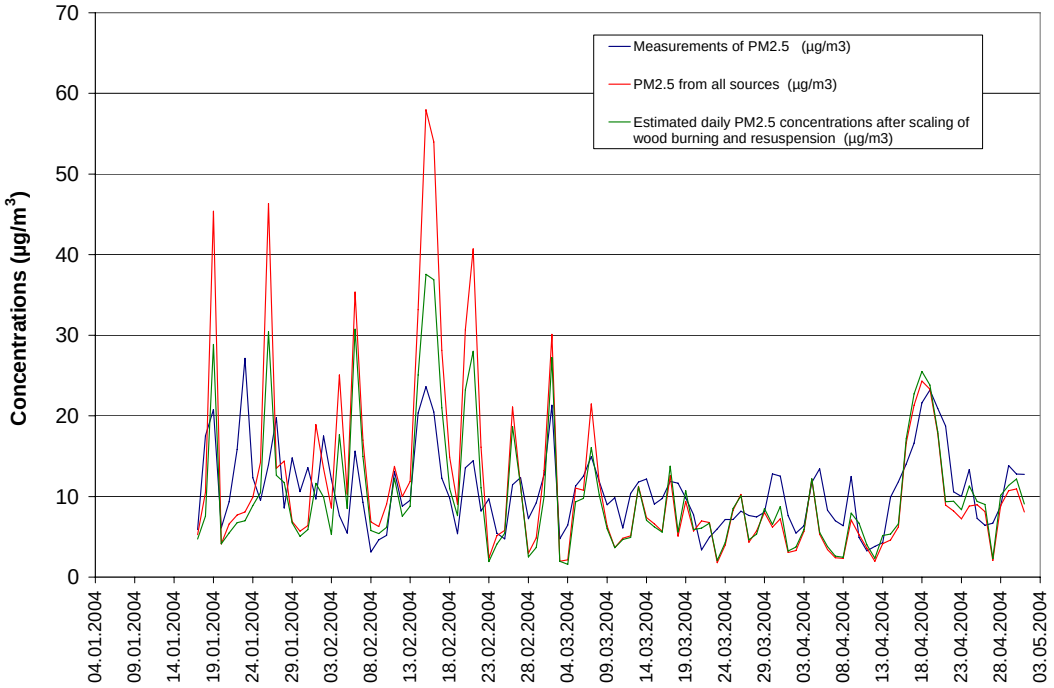

Figure 25: Estimated and measured daily $PM_{2.5}$ at Aker Hospital 16.01.2004- 1.05.2004, before and after scaling of wood burning and re-suspension.

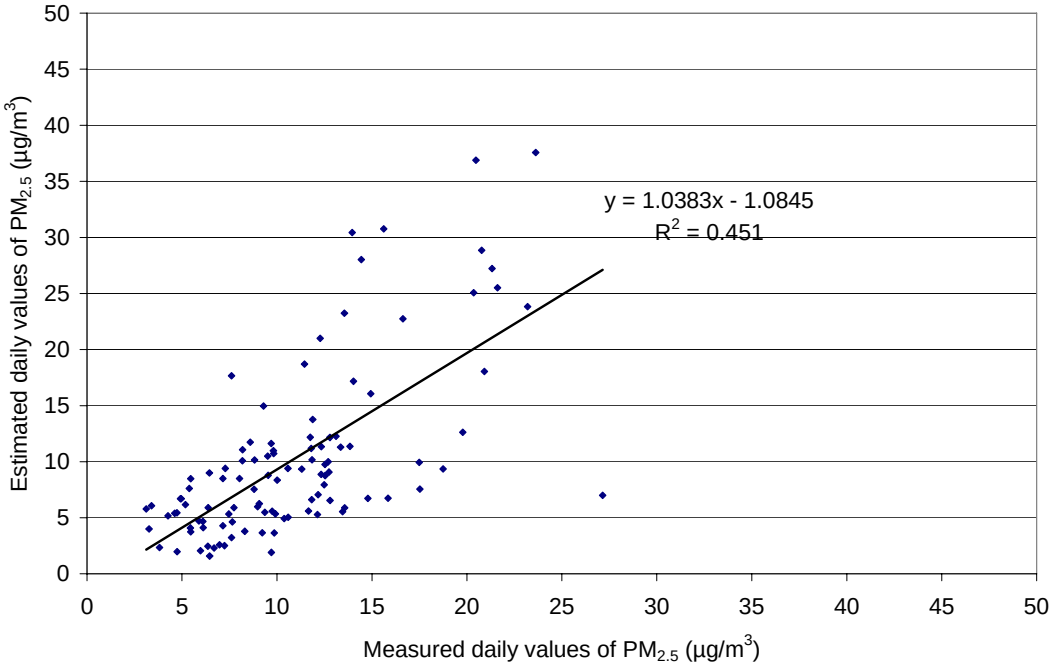

Figure 26: Estimated and measured daily $PM_{2.5}$ at Aker Hospital 16.01.2004- 1.05.2004 after scaling of wood burning and re-suspension.

Appendix C

Methods for calculating uncertainties parameters

RME and RPE as an uncertainty parameter

The Relative Maximum Error without timing (RME), which is the largest concentration difference of all percentile (p) differences normalized by the respective measured value is defined by: Co_p and Cp_p are the concentration observed and predicted values at the percentile (p).

$$RME = \frac{\max(\left|Co_p - Cp_p\right|)}{Co_p}$$

and when applied to annual means this indicator is almost the same as the absolute relative bias of a model in regard to observations. It is defined by

$$NME_i = \left|\frac{Cp_i - Co_i}{Co_i}\right|$$

Defining the quality indicator as the concentration difference at the percentile corresponding to the allowed number of exceedances of the limit value normalized by the observation (Relative Percentile Error - RPE).

$$\mathrm{RPE} = \frac{\left|\mathrm{Co_p} - \mathrm{Cp_p}\right|}{\mathrm{Co_p}}$$

Indicative RMSE

An uncertainty estimates can be produced by calculating the root mean square error (RMSE) of the model based on the available observations.

$$RMSE = \sqrt{\frac{1}{n}\sum_{i=1}^{n}\left(Cp_i - Co_i\right)^2}$$

By normalising this with the mean of the model concentrations at the observational points, the normalised RMSE (NRSME) can be determined

$$NRMSE = \frac{RMSE}{\overline{Cp_i}}$$

Appendix D

Statistical analysis for PM2.5

Table 6: Measured and estimated daily values (24 hours) of $PM_{2.5}$ at Kirkeveien, 01.11.2003-01.05.2004.

	Kirkeveien	Kirkeveien	Kirkeveien
	Measured $PM_{2.5}$,(µg/m^3), TEOM	**Estimated $PM_{2.5}$ (µg/m^3)**	**Estimated daily $PM_{2.5}$ concentrations after re-calculation (µg/m^3)**
Maximum	36.38	71.72	47.45
Average	13.59	14.08	11.75
Std. deviation	5.73	10.74	7.25
Correlation		0.66	0.72
Slope		1.24	0.91
Intercept		-2.81	-0.59
RMSE		8.15	5.39
NRMSE		0.58	0.46
RME*		0.04	0.14
RME		1.07	0.51
RPE_{36}		0.12	0.08
RPE_{8}		0.52	0.16
$RPE_{Average}$ (180 daily values)		0.20	0.21

RMSE: Root Mean Square Error, NRMSE: Normalized Root Mean Square Error, RME: Relative Maximum Error without timing. *This parameter is calculated based on the half year average value. RPE_{36}: Relative percentile error for the 36th highest daily value. RPE_8: Relative percentile error for the 8th highest daily value

Table 7: Measured and estimated daily values of $PM_{2.5}$ at Løren, 01.11.2003-01.05.2004.

	Løren	Løren	Løren
	Measured $PM_{2.5}$,(µg/m^3), TEOM	**Estimated $PM_{2.5}$ (µg/m^3)**	**Estimated daily $PM_{2.5}$ concentrations after re-calculation (µg/m^3)**
Max	31.78	61.02	45.37
Average	15.33	14.16	14.52
Std. deviation	6.49	10.63	9.15
Correlation		0.59	0.67
Slope		0.96	0.94
Intercept		-0.56	0.06
RMSE		8.62	6.84
NRMSE		0.61	0.47
RME *		0.076	0.053
RME		0.92	0.59
RPE_{36}		0.09	0.01
RPE_{8}		0.43	0.23
$RPE_{Average}$ (180 daily values)		0.26	0.20

RMSE: Root Mean Square Error, NRMSE: Normalized Root Mean Square Error, RME: Relative Maximum Error without timing. *This parameter is calculated based on the half year average value. RPE_{36}: Relative percentile error for the 36th highest daily value. RPE_8: Relative percentile error for the 8th highest daily value

Table 8: Measured and estimated daily values of $PM_{2.5}$ at Rv4, 08.01.2004-01.05.2004

	RV4	RV4	RV4
	Measured $PM_{2.5}$,(µg/m^3), TEOM	**Estimated $PM_{2.5}$ (µg/m^3)**	**Estimated daily $PM_{2.5}$ concentrations after re-calculation (µg/m^3)**
Max	29.82	60.27	40.62
Average	13.02	14.27	14.19
Std. deviation	5.25	10.96	7.95
Correlation		0.55	0.65
Slope		1.14	0.98
Intercept		-0.62	1.45
RMSE		9.25	6.16
NRMSE		0.65	0.43
RME*		0.096	0.090
RME		1.31	0.67
RPE_{36}		0.64	0.35
RPE_{8}		0.02	0.08
$RPE_{Average}$ (114 daily values)		0.21	0.11

Note: At Rv4, $PM_{2.5}$ were only measured from 08.01.2004-01.05.2004 RMSE: Root Mean Square Error, NRMSE: Normalized Root Mean Square Error, RME: Relative Maximum Error without timing. *This parameter is calculated based on the half year average value. RPE_{36}: Relative percentile error for the 36th highest daily value. RPE_8: Relative percentile error for the 8th highest daily value.

Table 9: Estimated and measured daily values of $PM_{2.5}$ at Aker Hospital, 16.01.2004-01.05.2004

	Aker Hospital	Aker Hospital	Aker Hospital
	Measured $PM_{2.5}$,(µg/m^3), TEOM	**Estimated $PM_{2.5}$ (µg/m^3)**	**Estimated daily $PM_{2.5}$ concentrations after re-calculation (µg/m^3)**
Max	27.14	57.98	37.57
Average	10.85	11.79	10.19
Std. Deviation	5.00	10.95	7.73
Correlation		0.61	0.67
Slope		1.34	1.04
Intercept		-2.80	-1.08
RMSE		8.82	5.74
NRMSE		0.75	0.56
RME		0.080	-0.062
RPE_{36}		1.29	0.56
RPE_{8}		0.11	0.18
$RPE_{Average}$ (180 daily values)		0.50	0.25
Relative Percentile Error Average (RPE) (106 daily values)		0.28	0.25

Note: At Aker hospital, $PM_{2.5}$ were only measured from 16.01.2004-01.05.2004 RMSE: Root Mean Square Error, NRMSE: Normalized Root Mean Square Error, RME: Relative Maximum Error without timing. *This parameter is calculated based on the half year average value. RPE_{36}: Relative percentile error for the 36th highest daily value. RPE_8: Relative percentile error for the 8th highest daily value.

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AUTHOR(S) Herdis Laupsa, Bruce Denby, Steinar Larssen and Jan Schaug	CLASSIFICATION * A CONTRACT REF.

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ABSTRACT
In the EU-funded project, Air4EU, a source apportionment study has been performed for Oslo. In this study, the quality of the emission inventory for particulates $PM_{2.5}$ was assessed. The assessment of the emission inventory was performed by comparing the estimated source contributions from the PMF model and the dispersion model.

NORWEGIAN TITLE

Estimat av kildebidrag til PM2.5 i Oslo fra spredningsmodellering og reseptormodellering.

KEYWORDS		
Source apportionment	PMF	Dispersion model

ABSTRACT (in Norwegian)

I EU- prosjektet, Air4EU, har NILU beregnet, ved bruk av en spredningsmodell og en reseptormodell, bidrag til PM2.5 fra ulike kilder i Oslo.

* Classification
- A Unclassified (can be ordered from NILU)
- B Restricted distribution
- C Classified (not to be distributed)