Air pollution at street level in European cities

European Environment Agency

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Executive summary

Traffic-related air pollution is still one of the most pressing problems in urban areas. Evidence of the adverse health effects of fine particulate matter is continuously emerging and it is alarming that most of the traffic-related emissions are in the fine particulates range (< $PM_{2.5}$). Human exposure to increased pollutant concentrations in densely populated urban areas is high. The improvement of air quality is therefore imperative. Air quality limit values, which are aimed at protecting public health, are frequently exceeded especially in streets and other urban hotspots.

This report studies the air pollution levels at traffic hotspot areas in 20 European cities compared to the urban background concentrations for NO_2, NO_X, PM_{10} and $PM_{2.5}$. To analyse and project air quality both the current situation (reference year 2000) and two scenarios aimed at 2030 (Current Legislation, CLE, and Maximum Feasible Reductions, MFR) were considered. The methodology applied in the report was developed in the ETC/ACC 'Street Emission Ceiling (SEC)' project. It aims to determine which local emission reductions are needed in streets in order to reach certain air quality thresholds. At its present stage of development, the SEC methodology allows analysis of air quality scenario projections at street level, and considers particular policies and measures at regional, urban and street scales.

Urban background concentrations were calculated for 20 European cities using the urban scale model OFIS. Regional background levels were derived from EMEP model results. For the reference year, the results of OFIS agree fairly well with corresponding Airbase measurement data. Reduced urban background air quality levels were obtained for both future scenarios studied. The largest improvement was for the MFR scenario.

Street increments (i.e. differences between street and urban background concentrations) were calculated using the street scale model OSPM. The modelled street increments vary from city to city because of street canyon geometry, wind direction and speed assumed. They are also defined by urban emission levels that lead to lower or higher urban background concentrations and by the vehicle fleet composition that gives lower or higher street scale emissions. Street level concentrations were calculated for three hypothetical street canyon configurations — wide, square and narrow. These are considered to represent a reasonable range of street canyon types across Europe. Assuming the same daily traffic load (20 000 vehicles per day) crossing the three types, the highest street increments are computed for the narrow canyon as its configuration leads to trapping of air pollutants inside the street.

Results for the reference year and a narrow canyon located in the centre of the city correspond well with observed street increments. The latter are found to decrease significantly in both scenarios; the maximum reduction resulting for the MFR scenario.

OFIS and OSPM model results were further analysed to discuss air quality limit value exceedances in the 20 European cities considered. Overall, the picture resulting for the narrow canyon situation in the reference year 2000 corresponds reasonably with the observations of both NO_2 and PM_{10}. The exceedance days calculated for PM_{10} in 2000 (according to the 2005 limit value, i.e. daily average of 50 $\mu g/m^3$ not to be exceeded more than 35 days a year) are higher than permitted in almost all cities in the narrow canyon, in 14 cities in the square canyon and in half the cities in the wide canyon case. It should however be noted that the aspect ratio considered for the wide canyon case is rather large and probably beyond the range of applicability of the OSPM model.

For the 2030 air quality projection, the results imply that at street level and for a narrow canyon the annual limit value ([1]) for NO_2 will be met in only very few cases for the CLE scenario and in most cases for the MFR scenario. However, the indicative limit value for PM_{10} is not expected to be met even in the MFR scenario. The permitted number of exceedances, according to the 2010 limit value, is expected to be met for NO_2 in all cities for the narrow canyon case including in the CLE scenario. However, exceedances of the PM_{10} indicative limit

([1]) According to Directive 1999/30/EC, in 2010 the limit values to be met for NO_2 are 40 $\mu g/m^3$ (annual average) and 200 $\mu g/m^3$ (hourly average not to be exceeded more than 18 times a year) whereas for PM_{10} the indicative limit values are 20 $\mu g/m^3$ (annual average) and 50 $\mu g/m^3$ (daily average not to be exceeded more that 7 days a year).

value are observed in certain cases including the MFR scenario. For $PM_{2.5}$ the reduction is in line with the significant reductions in the urban and in the street scale PM emissions attributed to the introduction of Euro V and Euro VI compliant vehicles.

Overall, the model results compare well with measurements, given the restrictions imposed by the similarity of the actual street canyon in which the measurements are made and the hypothetical street canyon configuration (traffic characteristics, street canyon location and geometry, etc.). For this reason, particularly unfavourable cases observed in certain cities, where exceptionally high concentrations are recorded, are difficult to model unless the specific street characteristics are known in detail. Detailed local traffic data combined with air quality measurements and data on the specific street are required in order to evaluate the overall methodology of this report. These are also necessary to determine the appropriateness of the selection of the particular street canyon configurations. The urban background concentrations produced with the available top-down emission inventories should be compared to up-to-date, bottom-up local emission inventories, where these are available. By doing this, local city development scenarios can also be evaluated. Finally, reliable vehicle fleets for new and non EU Member States are required in order to obtain accurate street level air quality projections for these cities, according to the latest version of TREMOVE.

1 Introduction

To assist the cost-effectiveness analysis of policy proposals for revised air quality legislation, the Clean Air for Europe programme (CAFE) specifically developed instruments combining state-of-the-art scientific models with validated databases which represented the situations of all Member States and economic sectors. The RAINS integrated assessment model was used to develop and analyse policy scenarios. The integrated assessment approach focuses on regional scale pollutant concentrations in Europe and primarily deals with long-range transport and the impact on vegetation and ecosystems. This is also in accordance with the analyses needed for the Convention on Long-range Transboundary Air Pollution. As ambient concentrations of certain air pollutants show strong variability at a much finer scale (e.g. urban and local scale), the CAFE programme also aims to address these air quality issues.

Within the framework of CAFE, the City-Delta project invited the scientific community to study the urban contribution to air pollution as estimated by regional scale models. The aim was to identify and quantify the factors that lead to systematic differences between urban and rural background air pollution concentrations. Useful functional relationships were developed within City-Delta which allow the determination of urban air quality levels as a function of rural background concentrations and local factors. As a limitation, however, these functional relationships are at present applicable only to the annual mean of the anthropogenic part of $PM_{2.5}$ (Cuvelier *et al.*, 2004). Funded by DG Research under the 5th Framework Programme, the MERLIN project studied the influence of effective regional air pollution abatement strategies to urban air quality, and how sufficient these may be in achieving compliance with both in-force and future limit values. The major contribution of urban emissions to urban scale pollution was confirmed which showed the need to address the design of air quality abatement strategies on an urban scale. The OFIS model was applied in the context of both the City-Delta and the MERLIN projects. This allowed for the assessment of the model's performance, while at the same time comparing the model results against measurements and the results of other models. The conclusion from both projects was that OFIS is a useful tool for investigating current and future air quality at the urban scale.

The basis for most current valid air quality standards are statistical correlations between the findings of epidemiological studies and measured urban background air pollution levels. Therefore, it should be considered as a success that current air quality assessment tools are capable of describing adequately urban background concentrations of regulated air pollutants. However, the majority of the urban population also spends a considerable amount of time in streets, which is a typical example of urban hotspots. Limit values also apply to these hotspots, where measurements across Europe show that air quality close to areas with increased traffic is of particular concern (e.g. EEA fact sheet TERM 04, 2004). Finer local-scale models are required to study air quality in streets. The work of van den Hout and Teeuwisse (2004) revealed the difficulty of classifying the various types of streets across European cities. Given that the particular hotspot characteristics significantly affect air pollutant concentrations, it considers the various street geometries and traffic parameters.

Since 2003, the European Environment Agency (EEA) has been funding the Street Emission Ceilings (SEC) project within the work programme of the European Topic Centre on Air and Climate Change (ETC/ACC). The main aim of SEC is to study street level air quality and to develop model assessment systems that may be used for integrated assessment purposes. At the same time, the study must also meet the needs of local authorities. Such systems should allow for the assessment of current air quality and future scenario projections, while considering focused policies and measures for the regional, urban and street scales (Annex A).

This report aims to use the expertise gained in SEC to provide an estimate of hotspot air pollution levels that occur at local scale within cities as compared to the urban background concentration levels. Annual NO_2, NO_X, PM_{10} and $PM_{2.5}$ values and daily or hourly exceedances are covered where applicable. Both the reference year situation and scenario projections are taken into account, while the multi-scale model application allows the description of the impact of particular policies and measures at the regional, urban and street scales. As an option, the approach suggested may be used to assess the effect of local measures on air quality at the urban and local scales.

The OFIS model was used to calculate urban background concentrations. The satisfactory performance of OFIS was demonstrated in the MERLIN and City-Delta projects and by the successful application of the EMEP/OFIS/OSPM sequence in SEC. The aforementioned limitations of the functional relationships developed in the City-Delta project were also taken into account.

2 Methodology

The methodology followed in calculating air pollution levels at hotspot areas across European cities largely follows the findings and the work performed during 2003–2004 in the ETC/ACC SEC project (Annex A). The work presented in this report follows the description included in the ETC/ACC 2005 Implementation Plan, task 4.4.1.3, 'Support of the CAFE programme regarding air pollution levels at hotspots'. Any additional details/clarifications were discussed with the CAFE Programme representatives.

Therefore, the methodology used to assess the impact of street scale emissions on the hotspot air pollution levels consists of:

(a) the urban scale — OFIS model (Arvanitis and Moussiopoulos, 2003). This is driven by results of the EMEP model (URL1) — concentrations and meteorological data — in order to obtain the urban background

(b) the local scale — OSPM model (Berkowicz *et al.*, 1997). This is driven by OFIS model results for estimating hotspot air pollution levels.

The results included in the report are for NO_2, NO_X, PM_{10} and $PM_{2.5}$. For the reference year, validation of model results has been performed against measurements available in Airbase (URL 2). Due to lack of sufficient data for certain cities and certain pollutants, data from the years 2001, 2002 and in some cases 2003 were used (see Annex B — additional details are available upon request). They represent good approximations for the level of the concentrations measured in 2000. For the projection of the street increments, a baseline (Current Legislation) and Maximum Feasible Reductions (MFR) scenario for the year 2030 are used. These are defined in Cofala *et al.* (2005).

Urban emission inventories were required as input for the OFIS model. A top-down approach was used with inventories developed in the MERLIN project for 20 cities ([2]). For local air quality analysis, specific street canyon characteristics were required in order to define particular case studies (types of streets) in each city. Due to the absence of such detailed data for street types across Europe, a generic approach was applied. The hypothetical street canyons for which the OSPM model was applied were defined from the 'Typology Methodology'. This represents a first attempt to categorise street types according to various parameters and parameter ranges (van den Hout and Teeuwisse, 2004). TREMOVE (De Ceuster *et al.*, 2005) and TRENDS (Giannouli *et al.*, 2005) models were used to calculate the vehicle fleet data, and local emissions are then calculated with the COPERT 3 emission model (Ntziachristos *et al.*, 2000).

Annual average concentrations and annual deltas (or 'street increments', i.e. the difference between the street and the urban background concentrations) were calculated for NO_2, NO_X, PM_{10} and $PM_{2.5}$ for the 20 cities. Hourly NO_2 and daily PM_{10} exceedances, as these are defined by the relevant legislation, were also calculated for the 20 cities. Based on the Typology Methodology report, hotspot air quality analysis was performed for the two specified urban canyon geometries (square and wide cases). In addition, a third geometry representing a narrow street canyon was also considered. The data available allowed for the analysis of a reference year (2000) and two alternatives for the year 2030: the Current Legislation and Maximum Feasible Reduction scenarios ([3]) described in detail elsewhere (Cofala *et al.*, 2005). As requested by CAFE representatives, compatibility with the TREMOVE model was ensured throughout the report and comparison of model results against observations is presented as far as possible.

([2]) Antwerp, Athens, Barcelona, Berlin, Brussels, Budapest, Copenhagen, Gdansk, Graz, Helsinki, Katowice, Lisbon, London, Marseilles, Milan, Paris, Prague, Rome, Stuttgart and Thessaloniki.

([3]) Assumptions on technologies adopted and efficiencies of control technologies in the MFR scenario are available from the RAINS website: http://www.iiasa.ac.at/web-apps/tap/RainsWeb/ under the scenario CP_MFR_Nov04(Nov04).

3 Emissions

Gridded urban emission inventories for the reference year 2000 were prepared by Stuttgart University, Institute of Energy Economics and the Rational Use of Energy (IER) within the framework of MERLIN, using the European Emission model (Friedrich and Reis, 2004; Schwarz, 2002; Wickert, 2001) The emission inventories were made available for the aforementioned 20 urban areas.

The urban emission projections for the year 2030 were predicted according to the emission control scenarios LGEP-CLE and LGEP-MFR (Cofala *et al.*, 2005). This gave appropriate sectoral emissions (Cofala, 2004). Since information of this type was only available at country level and not at city level, the emission reductions were calculated for each country (Austria, Belgium, Czech Republic, Denmark, Finland, France, Germany, Greece, Hungary, Italy, Poland, Portugal, Spain, United Kingdom), SNAP category (SNAP 1 to 10 as described in Annex C, table C1) and pollutant (NO_X, VOC, SO_2, NH3, PM_{10} and $PM_{2.5}$) for the year 2030. The emission reductions at urban level were then considered equal to those at country level. This gave the urban emissions per pollutant and SNAP category for the year 2030. Details on the methodology followed may be found in Annex C.

Vehicle fleets extracted from TRENDS (Giannouli *et al.*, 2005) and TREMOVE (De Ceuster *et al.*, 2005) models were used in order to calculate reference year local (street) emissions with COPERT (Ntziachristos *et al.*, 2000) for a narrow street canyon. A narrow street canyon was assumed to have an average daily traffic of 20 000 vehicles (see Annex C, table C4). Generic values were used for the remaining parameters (vehicle speed, percentage of heavy-duty vehicles in the fleet — henceforth: HDV % —, street canyon geometry etc.). For consistency reasons, these values were assumed to coincide with those defined in the Typology Methodology for urban canyons (van den Hout and Teeuwisse, 2004). The methodology adopted for the calculation of local scale emissions is further described in Annex C of this report.

4 Urban and local scale air quality

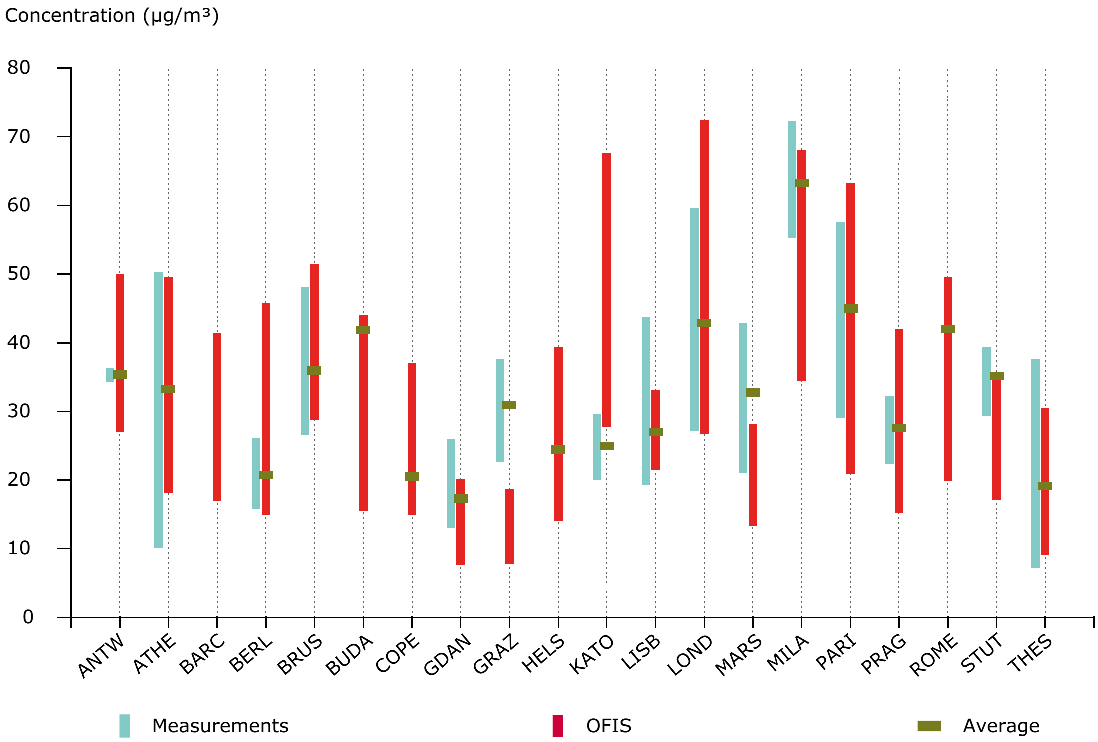
In this section, current and future air quality at urban and street scale in 20 European cities is investigated in terms of the annual mean concentrations for NO_2, NO_X, PM_{10} and $PM_{2.5}$, and exceedances of the hourly and daily 2010 limit values for NO_2 and PM_{10} respectively. The model simulations were performed with the multi-scale model cascade EMEP/OFIS/OSPM (Arvanitis and Moussiopoulos, 2003; Berkowicz *et al.*, 1997). This approach allows a complete analysis of both the reference year situation and scenario projections as the impact of air pollution control strategies and measures are accounted for at all relevant scales (regional, urban and street scale).

4.1 Reference year (2000) and validation against measurements

4.1.1 Urban air quality

In Figures 4.1 to 4.5 OFIS model results for the reference year 2000 are compared to Airbase data for NO_2, NO_X, PM_{10} and as far as possible $PM_{2.5}$ using urban and suburban background station measurements. To account for the variability in the background concentrations in each city, the figures show the ranges for both observations and model results. As expected, the model predicts maximum values for all pollutants (NO_2, NO_X, PM_{10} and $PM_{2.5}$) in the city centre. For cities where there is only one station available, it is not possible to define such a range. Furthermore, the concentration observed at the particular location should be treated as indicative. The appropriateness of the reported background concentrations depends upon the number and types of stations in each city. The issue of 'how well they represent population exposure' should also be considered. In Figures 4.1 to 4.4 the average value of all stations in each city (noted as average in the graphs) is also shown for comparison. A full list of stations used in this analysis can be found in Annex B.

Figure 4.1 Mean annual NO_2 urban background concentrations (µg/m³) in 20 European cities: range of OFIS model results for the reference year 2000 compared to the range of observations and average value of all stations



For the NO_2 concentrations, there is clear agreement between OFIS model results and urban background measurements. The spread of the OFIS values mostly overlaps the spread in the measured data, though in some cases the maximum value is overestimated by the model. Good agreement with measurements is also obtained in the case of NO_x, though in some cases an underestimation is observed. OFIS generally refines the regional model results, thus leading to a better estimate of the urban background NO_2 and NO_x concentrations. As an exception to this very satisfactory general agreement, a large discrepancy between model results and observations is detected for Graz and Marseilles (Figure 4.1). This is due to an underestimation of the urban NO_x emissions which results from the application of a top-down approach (from NUTS 3 down to the domain of interest) of the European emission model (Friedrich and Reis, 2004; Schwarz, 2002; Wickert, 2001). The European emission model produces gridded emission inventories. A better result would have occurred for the emission inventory if a bottom-up approach (emission inventory using local data) had been used.

Figure 4.2 Mean annual NO_x urban background concentrations (µg/m³) in 20 European cities: range of OFIS model results for the reference year 2000 compared to the range of observations and average value of all stations

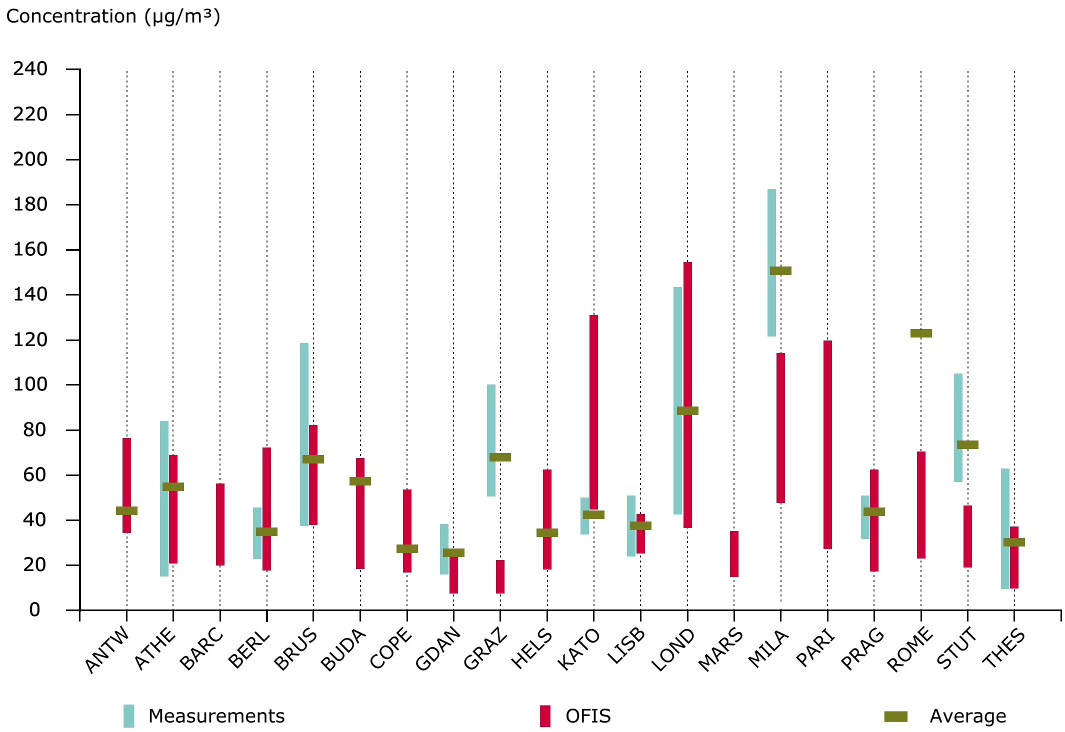


Figure 4.3 Mean annual PM_{10} urban background concentrations (µg/m³) in 20 European cities: range of OFIS model results for the reference year 2000 compared to the range of observations and average value of all stations

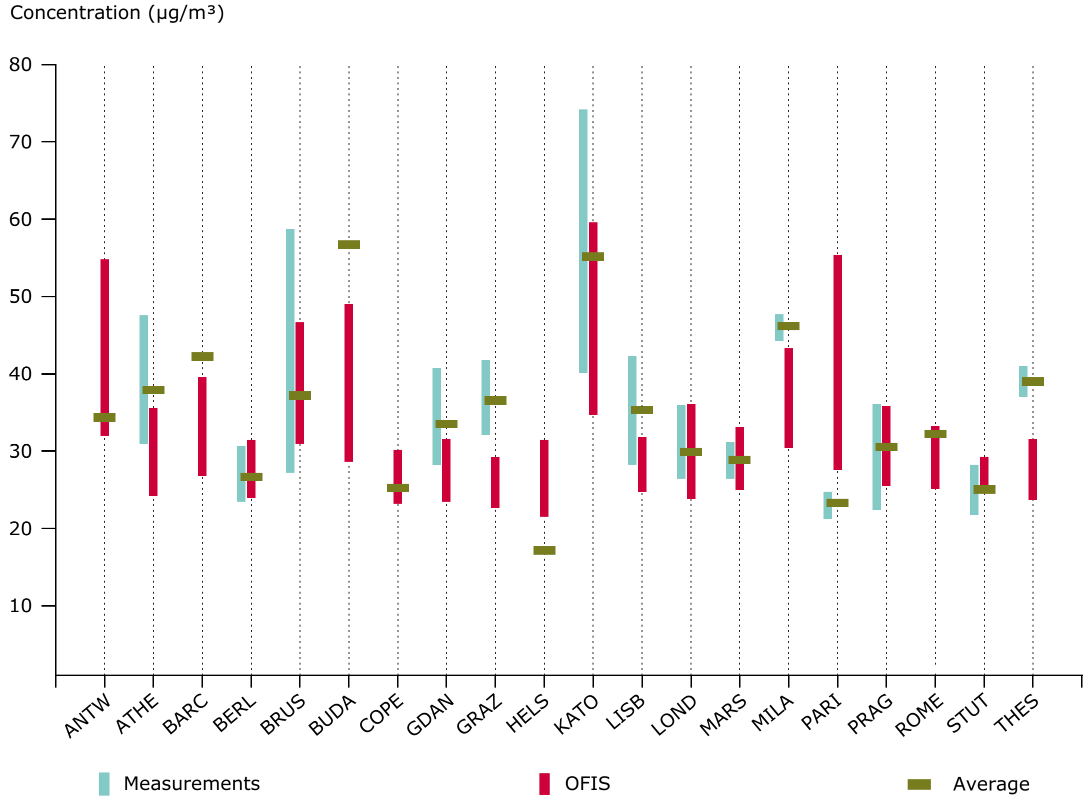
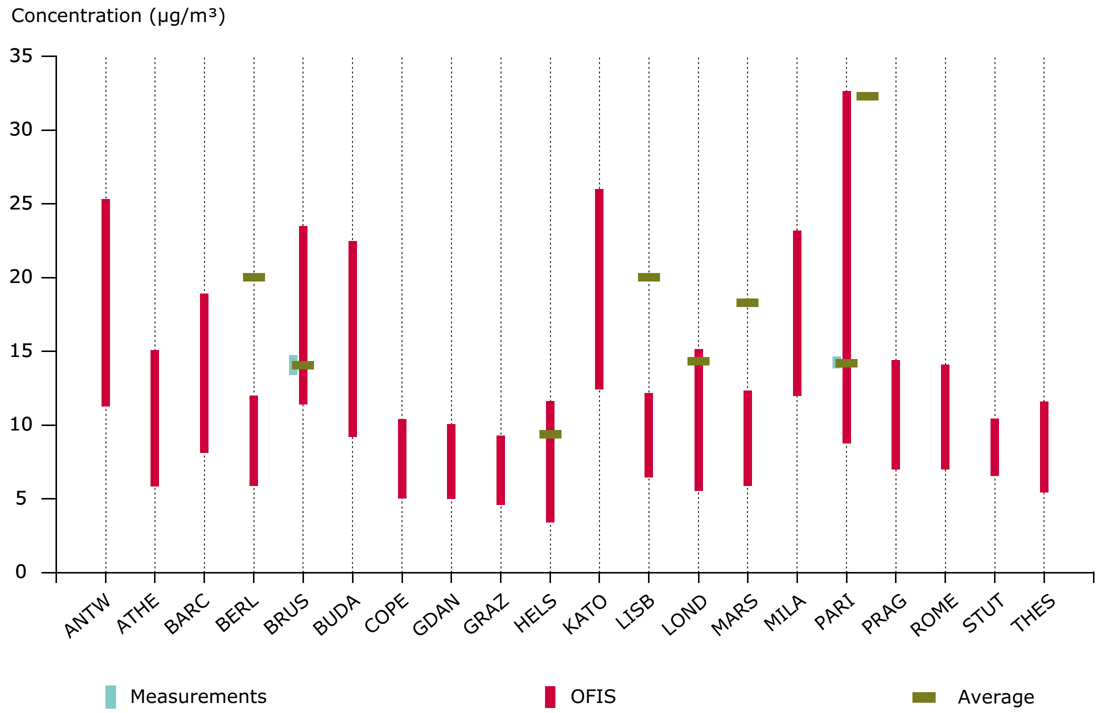


Figure 4.4 Mean annual $PM_{2.5}$ urban background concentrations (µg/m³) in 20 European cities: range of OFIS model results for the reference year 2000 compared to the range of observations and average value of all stations



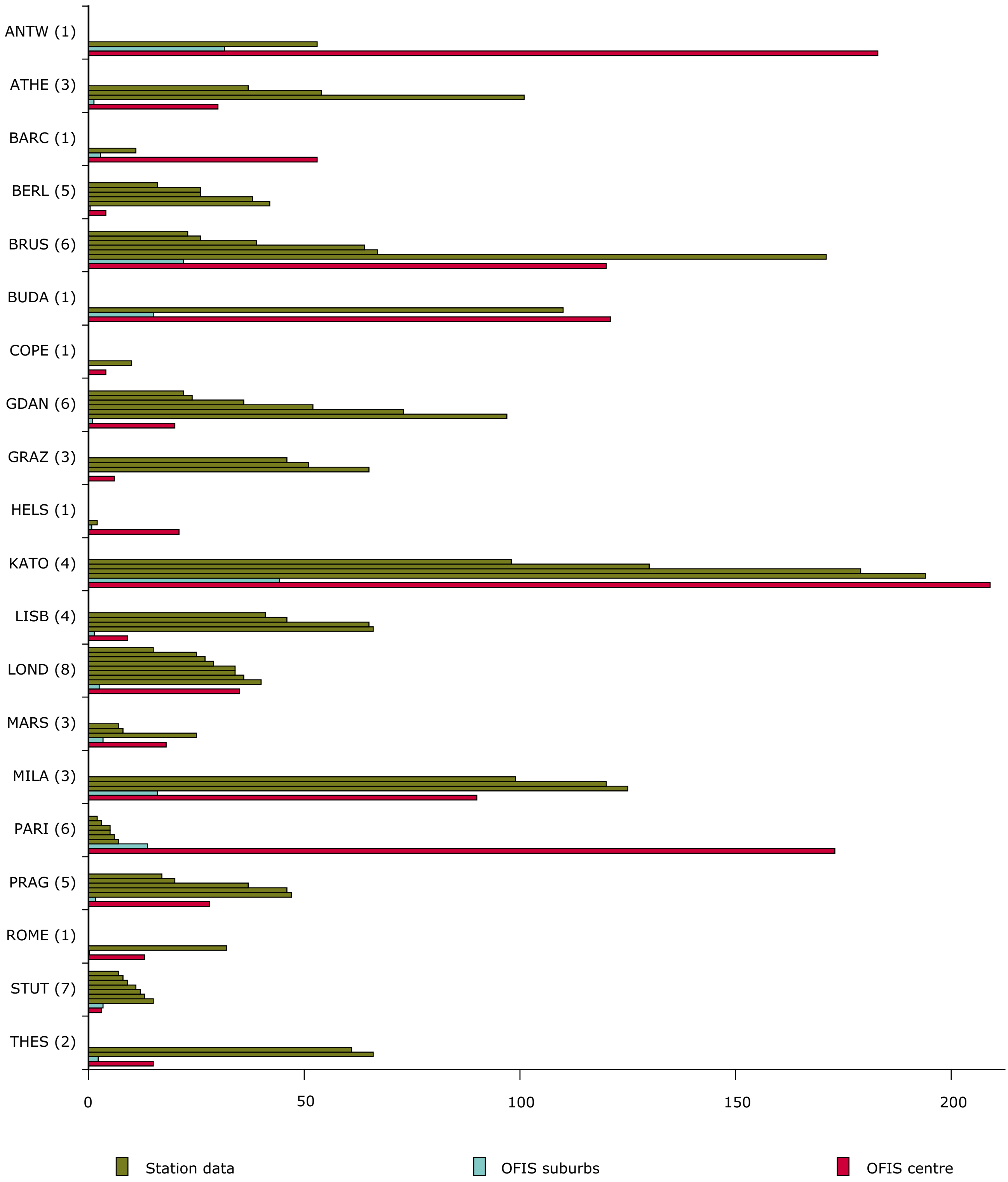
For PM_{10}, a reasonably good comparison with measurements is achieved. As neither the regional (EMEP) nor the urban scale (OFIS) model accounts for natural primary PM sources, such as windblown dust (African dust and local soil resuspension), sea salt or organic aerosols, a constant value of 17 $\mu g/m^3$ has been assumed for all cities to account for these PM sources. The value was estimated as an average across all data available for the annual mean PM_{10} concentration measured at the EMEP Measurement network stations (28 stations in 2001, 30 stations in 2002) (URL3). It should be noted that these stations are unevenly located across Europe since there are many countries with no data. Therefore, this estimate may either overestimate or underestimate natural sources in some cases. For example, it should perhaps be larger in the case of cities located in dry costal areas of Southern Europe where PM sources such as African dust, local soil resuspension and sea salt would make a larger contribution, Similarly, this should be the case for coastal cities in Northern Europe where sea salt would again play an important role in PM_{10} concentrations. Overall, it must be noted that primary PM_{10} emission data are not as robust as those for other air pollutants. This, combined with the complex formation, deposition and resuspension processes, leads to uncertainties for the modelled PM_{10} ambient concentrations. Also, OFIS, like many urban scale models, does not yet account for the formation of secondary organic particulates. This is an omission that could lead to an underestimation of the modelled PM_{10} concentrations.

For $PM_{2.5}$ there are very few measurements to validate the model results. In cases such as Brussels, Helsinki, London and Paris the limited data are found to be within the range of the model results. However, in cases such as Berlin, Lisbon and Marseilles an underestimation is observed. A possible reason for this is that the formation of secondary organic particulates is not accounted for by OFIS.

In Figure 4.5 the number of exceedances of the daily PM_{10} limit value (50 $\mu g/m^3$) has been computed. The constant value of 17 $\mu g/m^3$ in the daily average model results has been included in the computation. The model results compare well with the measured data. The overestimation or the underestimation of the number of exceedances in most cases clearly follows the overestimation or underestimation observed in the annual mean concentration results (see Figure 4.3). Although it seems reasonable to add a constant value of ~ 17 $\mu g/m^3$ to the annual mean PM_{10} model results, the constant value needed to be added to the daily average model results in order to calculate exceedance days is a more complex issue. This constant value will vary largely from city to city depending on its location (e.g. southern/northern Europe, coastal or non-coastal city) and season (e.g. windy summer days). This gives an uncertainty of perhaps ± 3–5 $\mu g/m^3$, which is considerable in view of the comparison with the limit value. The variation of PM_{10} concentrations across Europe is obviously an important scientific issue and deserves special attention. However, this goes beyond the scope of the report. Despite the limitations of the approach followed in this analysis, Figure 4.5 still provides a useful insight into the amount of exceedances in cities across Europe.

Exceedances above the hourly NO_2 limit value for 2010 (200 $\mu g/m^3$) are rarely observed in the urban and suburban background station data and the urban scale model results. When they are observed, they tend to be below the allowed number of exceedances (18 times a year). Therefore, this comparison is only presented for the traffic station data and OSPM model results (see Section 4.1.2).

Figure 4.5 Number of daily exceedances of the 50 µg/m³ limit value for PM_{10} in 20 European cities: OFIS model results for the city centre and the suburbs compared to observations



Note: The number of urban background stations available in each city is noted in brackets.

4.1.2 Local air quality

The NO_2, NO_X, PM_{10} and $PM_{2.5}$ concentrations measured at urban traffic stations across Europe are higher than those at urban background stations. This is due to increased local emissions from road traffic. The concentrations measured at traffic stations largely depend on a number of factors, namely: the specific street configuration; the traffic characteristics; the orientation of the street with respect to the prevailing wind direction; the location of the street and the location of the traffic station in the street itself. Hence, it is difficult to define a representative range of values. For the same reasons, the concentrations modelled will largely depend on the specific street configurations considered and also the HDV % and the average vehicle speed assumed. These considerations are the most important parameters governing the street emissions.

In the analysis that follows, the streets were assumed to be centrally located, i.e. the urban background concentrations were assumed to be adequately described by the OFIS model results for the centre of the city. The street orientation was assumed to be 'east to west', and the wind speed and direction for each city were derived from the EMEP data. The yearly average wind speeds for each city can be found in Annex D. For quantifying the hotspot contributions, it is convenient to introduce street increments, i.e. the difference between the street and the urban background concentrations. Model results are presented, and street increments comparison against measurements is performed.

The measured street increments were calculated using the maximum measured street and background concentrations in each city. These were considered to represent as far as possible the concentrations observed close to the centre of the city, and so were comparable to the modelled street increments. Inevitably, this introduces an uncertainty since the increment depends critically on the location of the respective urban background and traffic stations, which are often not close to each other. This can lead to either an overestimation or an underestimation of the street increments depending on whether the street station is located in the city centre and the urban background station is far from the centre or vice-versa. Moreover, agreement or disagreement between measured and modelled street increments will be strongly affected by the question of how similar the actual street geometry, orientation, traffic characteristics etc. are compared to the hypothetical streets studied. Answering this question, however, would have required a detailed analysis of the characteristics of the street canyons where the traffic stations operate; a task well beyond the scope of the present study.

Street increments for NO_2, NO_X, PM_{10} and $PM_{2.5}$ were calculated with the OSPM model for three hypothetical street canyon configurations. The square (height and width = 15 m) and wide (height = 15 m, width = 40 m) canyons were defined according to van den Hout and Teeuwisse (2004). The third canyon was selected to represent a narrow canyon case (height = 15 m and width = 10 m). It was assumed that the number of vehicles crossing each type of canyon and corresponding emissions would differ depending on the canyon width. It was also the assumption that the narrow canyon had 20 000 vehicles per day, the square 30 000 vehicles per day and the wide 60 000 vehicles per day.

As expected, the differences between the street increments computed for the three canyon geometries are generally small. In most cases the largest increments are observed for the wide canyon due to the increased number of vehicles, and hence the emissions that lead to high street-level concentrations within this canyon. It should, however, be noted that the aspect ratio of the wide canyon case (2.7), following van den Hout and Teeuwisse (2004), is rather large. Thus, the applicability of the OSPM model is doubtful. The results of the modelled against the measured street increments for the narrow canyon case and for the reference year (2000) are presented in Figures 4.6 to 4.9. The hourly NO_2 and daily PM_{10} exceedances for the narrow case are also shown in Figures 4.10 and 4.11. Details concerning the calculations of the street emissions can be found in Annex C. Here, the methodology is analysed and the emissions for the narrow canyon with 20 000 vehicles per day are presented. These differ from city to city according to the specific fleet composition and contribution of each vehicle category to the total street emissions. The HDV % and the average vehicle speed (26 km/h) used for the emission calculations were defined by the Typology Methodology report (van den Hout and Teeuwisse, 2004). The report foresees one of two discrete values (7 % or 15 %). Based on TRENDS/TREMOVE model results for the country scale, the larger value was used only for Lisbon.

In order to study the street increment sensitivity to an increased HDV %, in Section 4.1.2.1, the narrow case results using 7 % HDV are compared to results using 15 % HDV for selected cities. Finally, in order to understand the influence of the different canyon geometries on the street level concentrations, OSPM model results were also computed for the three canyon types. Here, the same number of vehicles per day (20 000) was assumed. The results for PM_{10} are shown in Figures 4.13 and 4.14, Section 4.1.2.2.

The aim of the calculations and the results presented in the figures below is not to show an ideal comparison with measurements. Due to the aforementioned constraints this is not possible. Instead, the aim is to provide an order of magnitude of the street increments for the various pollutants across European cities.

Figure 4.6 Mean annual NO_2 street increments (μg/m³) for the reference year 2000 in 20 European cities: model results for the narrow canyon case compared to observations

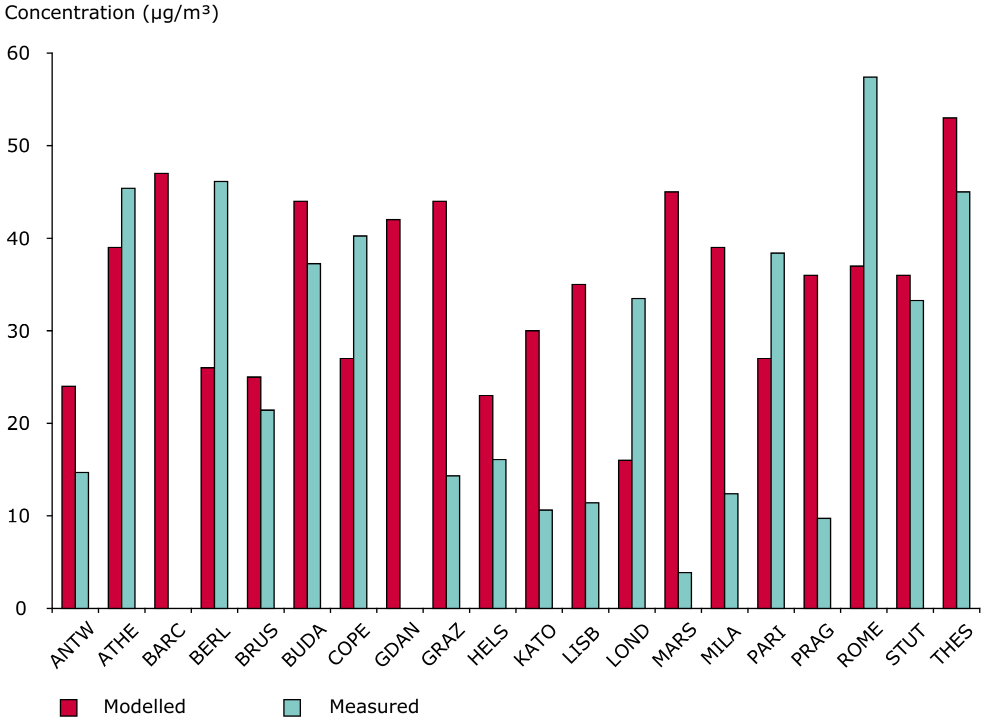
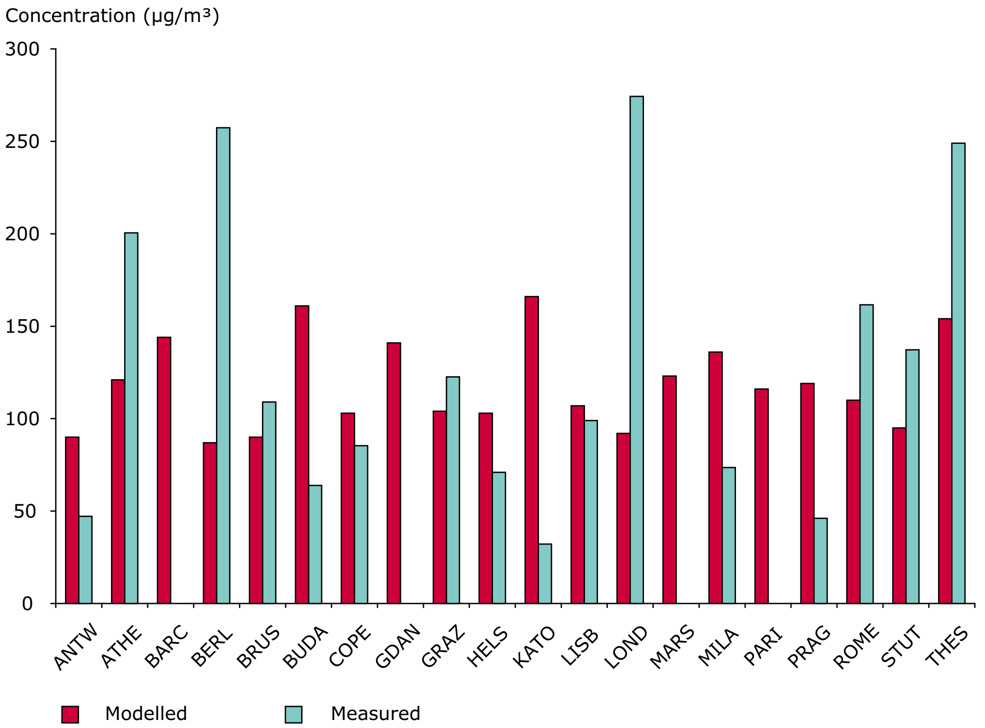


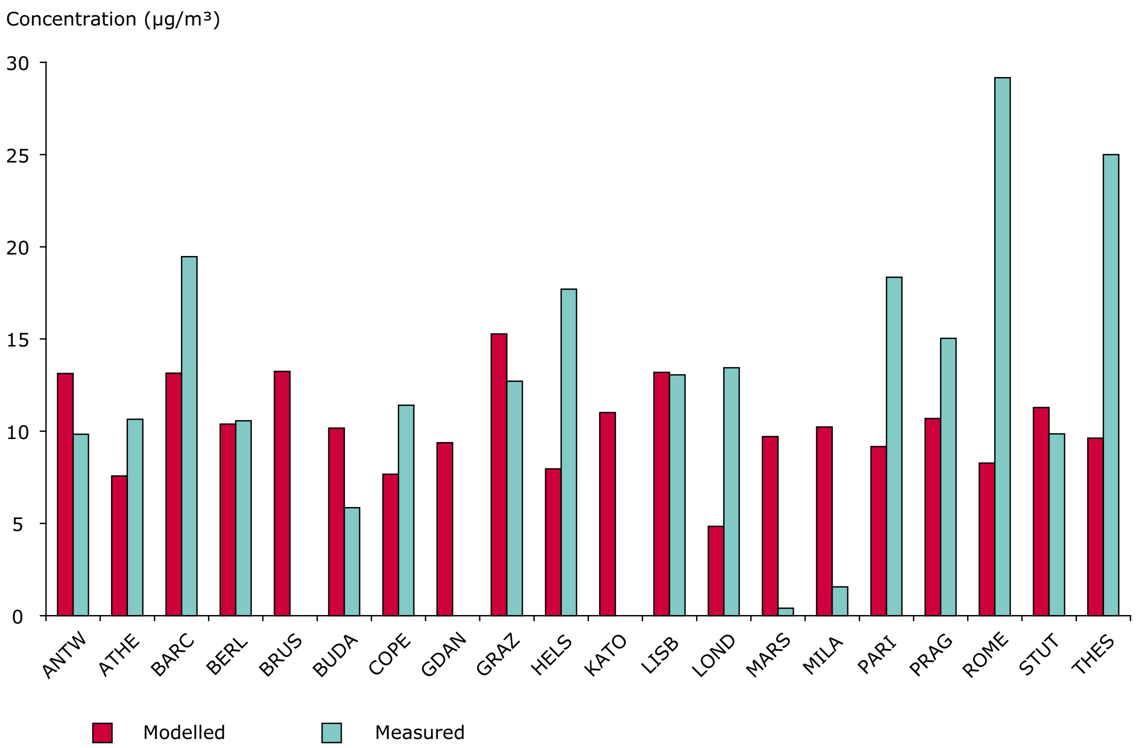
Figure 4.7 Mean annual NO_x street increments (μg/m³) for the reference year 2000 in 20 European cities: model results for the narrow canyon case compared to observations



For the narrow street canyon, large but comparable variations of the measured and the modelled street increments of NO_2 (10–57 μg/m^3 and 16–53 μg/m^3 respectively) are observed from city to city. In the case of Marseilles, an unrealistically low street increment (to be considered representative for the whole city) of 4 μg/m^3 is observed. This could be due to the high concentration recorded at the background station or to the low concentration recorded at the traffic station. However, detailed information on the exact station location would be required in order to draw conclusions on the representativeness of these stations. In the case of NO_X, the range of the measured street increments varies significantly. A lower than expected street increment is calculated in some cases due to unrealistically low traffic station measurements, such as the case of Katowice. Here, the traffic station is located outside the urban core and hence is not representative of the concentrations measured at traffic stations inside Katowice. In other cases, such as Berlin, London and Thessaloniki, an exceptionally high traffic measurement is recorded which gives a large measured street increment. The modelled increment range is 87–166 μg/m^3 whereas the measured range is 32–275 μg/m^3.

For PM_{10} the range of the modelled street increments in the narrow street canyon is 5–15 μg/m^3. The average value is 10 μg/m^3. The average value of the measured street increments from the stations in Figure 4.8 (as many station pairs as possible, not considering their proximity) is 13 μg/m^3. However, if the exceptionally large increments in Rome and Thessaloniki are not considered, this drops to 11 μg/m^3. These large increments appear to be due to exceptionally high concentrations measured at traffic stations. However, this issue cannot be studied further as details on the precise street canyon configurations are not available. In analyses conducted using 16 station pairs (traffic and urban background station pairs) for 2002 and for stations located close to each other (i.e. less than 1 km apart) the annual mean PM_{10} street increment was found to be 6.9 μg/m^3 (EEA, 2005b). Bearing in mind all the limitations associated with the comparison of measured and modelled street increments, the modelling approach seems to reproduce the observed PM_{10} concentrations fairly well.

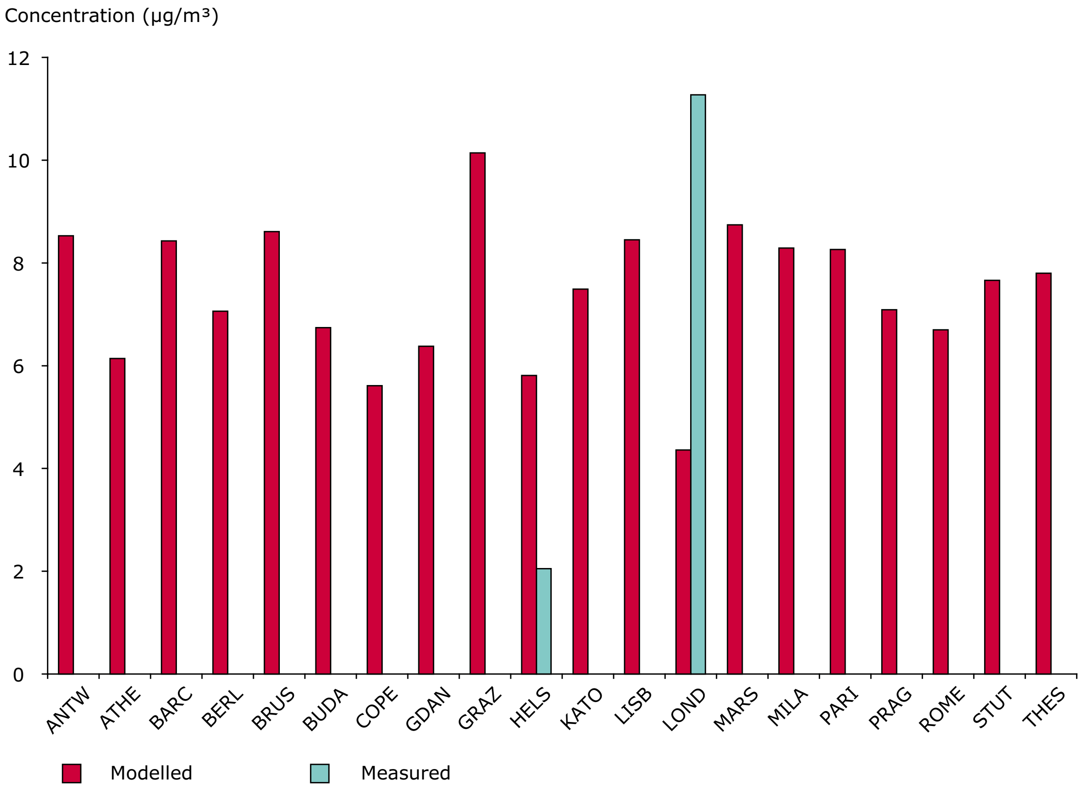
Figure 4.8 Mean annual PM_{10} street increments (μg/m^3) for the reference year 2000 in 20 European cities: model results for the narrow canyon case compared to observations



For $PM_{2.5}$ the range of the modelled street increments for the narrow canyon is 4–10 µg/m^3. From the limited data available, the measured increment is found to range from 2 µg/m^3 in Helsinki to 11.3 µg/m^3 in London. In the case of London, the street increment is calculated using the traffic station located at Marylebone Road and the urban background station at Bloomsbury. The corresponding modelled increment for London for the wide canyon is ~ 4 µg/m^3. For Marylebone, the difference between these two values can be attributed to an underestimation of the street level concentrations since the urban background measurements correspond well with the model results (see Figure 4.4 and corresponding analysis). The modelled street concentrations may have been underestimated since the actual HDV % of Marylebone is 10 %, whereas the hypothetical street canyon assumes 7 %, and also Marylebone has much more traffic (~ 85 000 vehicles per day) than that assumed in the wide canyon case (60 000 vehicles per day).

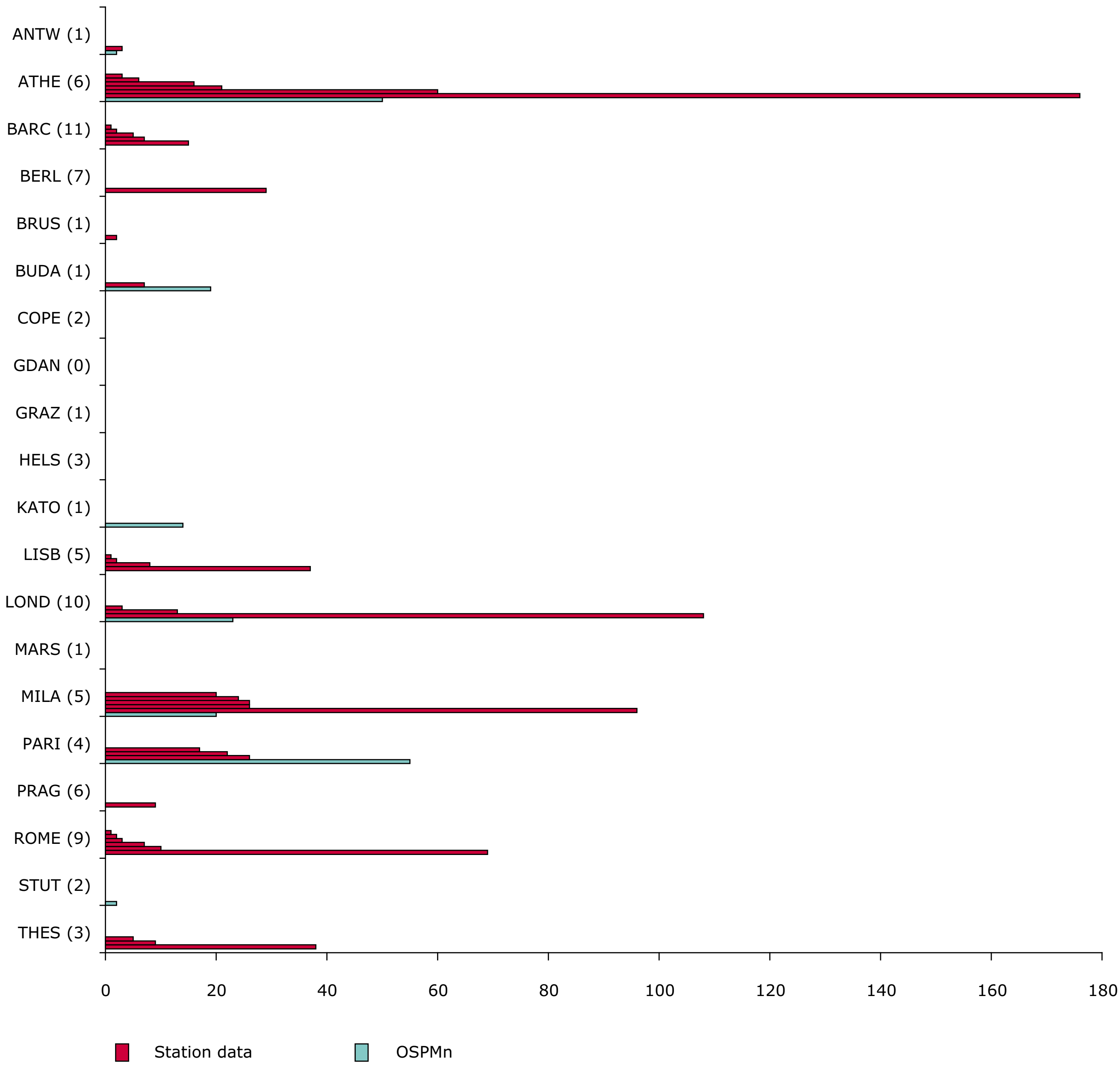
Overall, the comparison of modelled street increments against measurements shows reasonable results. However, one has to bear in mind all the limitations associated with this comparison. These limitations include the actual distance between the location of the traffic and urban background stations, their distance from the city centre and the differences in the street canyon geometries considered. It is apparent that a measured increment exceeding the modelled one could be associated with the use of a much too low urban background value. On the other hand, the opposite could well imply that the actual highest traffic concentrations in the city exceed the measured street concentrations. Also, in terms of the model results and assumptions, it is likely that the average vehicle speed of 26 km/h considered following van den Hout and Teeuwisse (2004) may be rather low. This could have led to slightly increased estimates of the exhaust PM emissions, and consequently an overestimation of the predicted concentrations. Furthermore, it is uncertain how accurately the non-exhaust PM_{10} and resuspension emissions were estimated (see Annex C). Depending on whether the PM emission sources are overestimated or underestimated, the corresponding PM_{10} street level concentrations will be affected. This would give a larger or smaller street increment respectively. Finally, the comparison also reveals the restrictions of the hypothetical street canyon configurations considered in this analysis. The worst street increments may have also been (see Rome and Thessaloniki PM_{10} street increments, Berlin, London and Thessaloniki NO_X street increments and London $PM_{2.5}$ increments) the worst street canyon configurations, i.e. the street geometry and traffic characteristics may not have been explicitly considered.

Figure 4.9 Mean annual $PM_{2.5}$ street increments (µg/m³) for the reference year 2000 in 20 European cities: model results for the narrow canyon case compared with observations



The hourly NO_2 and daily PM_{10} exceedances at street level were also calculated using the OSPM model for the three different street configurations. In Figures 4.10 and 4.11 the model results are compared to measured exceedances observed at various traffic stations across each city.

Figure 4.10 Number of hourly NO_2 exceedances of the 200 µg/m³ limit value in 20 European cities for the narrow canyon case



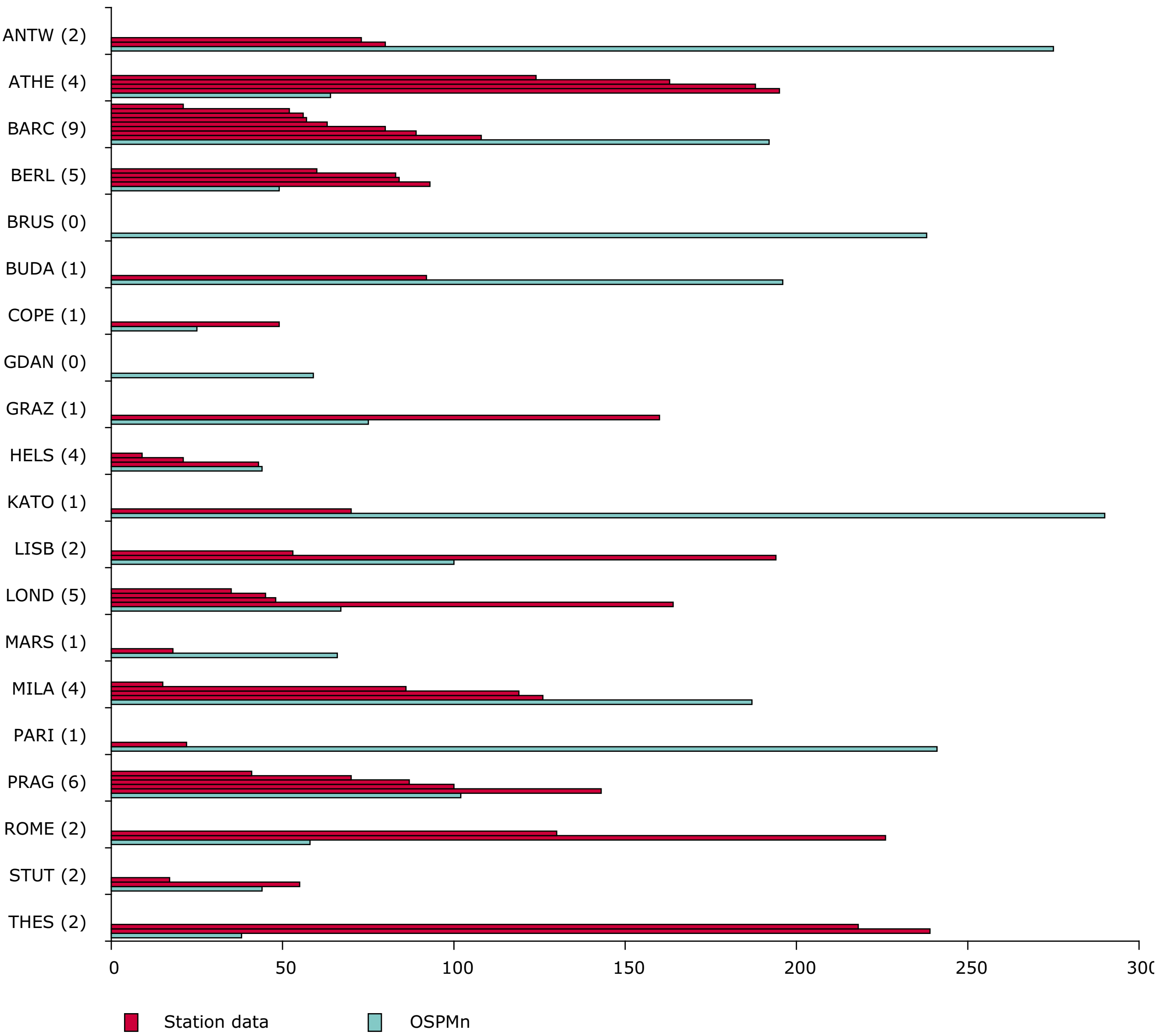
Note: The number of urban traffic stations available in each city is noted in brackets.

The exceedance results for both NO_2 and PM_{10} are reasonably good. However, the exceptionally high exceedances observed at specific stations (worst cases) cannot be modelled, since (as was also noted in the street increment analysis) the worst street canyon cases have not been considered. For PM_{10} the overall under-estimation or over-estimation of the exceedances observed for certain cities (Antwerp, Athens, Graz, Paris) follows from the over-estimation or under-estimation of the urban background concentrations (OFIS results). These were requested as input by the street scale model OSPM (see also Figure 4.3) since they play an important role in the concentrations computed at street scale. In cities such as Berlin, Copenhagen and Prague, where there is fair agreement between modelled and measured urban background levels (Figure 4.3); the exceedances calculated at street level are also in agreement with the exceedances measured at the various traffic stations. Overall, the accuracy of the modelled exceedances appears to be very sensitive to the accuracy of the modelled annual mean concentrations.

4.1.2.1 *The influence of an increased HDV %*

In order to study the street increment sensitivity to the HDV %, the street emissions for Athens, Berlin, Milan, Rome, Stuttgart and Thessaloniki were also computed based on an HDV % of 15 %. In Figure 4.12 the street increments corresponding to these emissions for the narrow street canyon with 20 000 vehicles per day are compared to the street increments for the same street canyon, but based on an HDV % of 7 %.

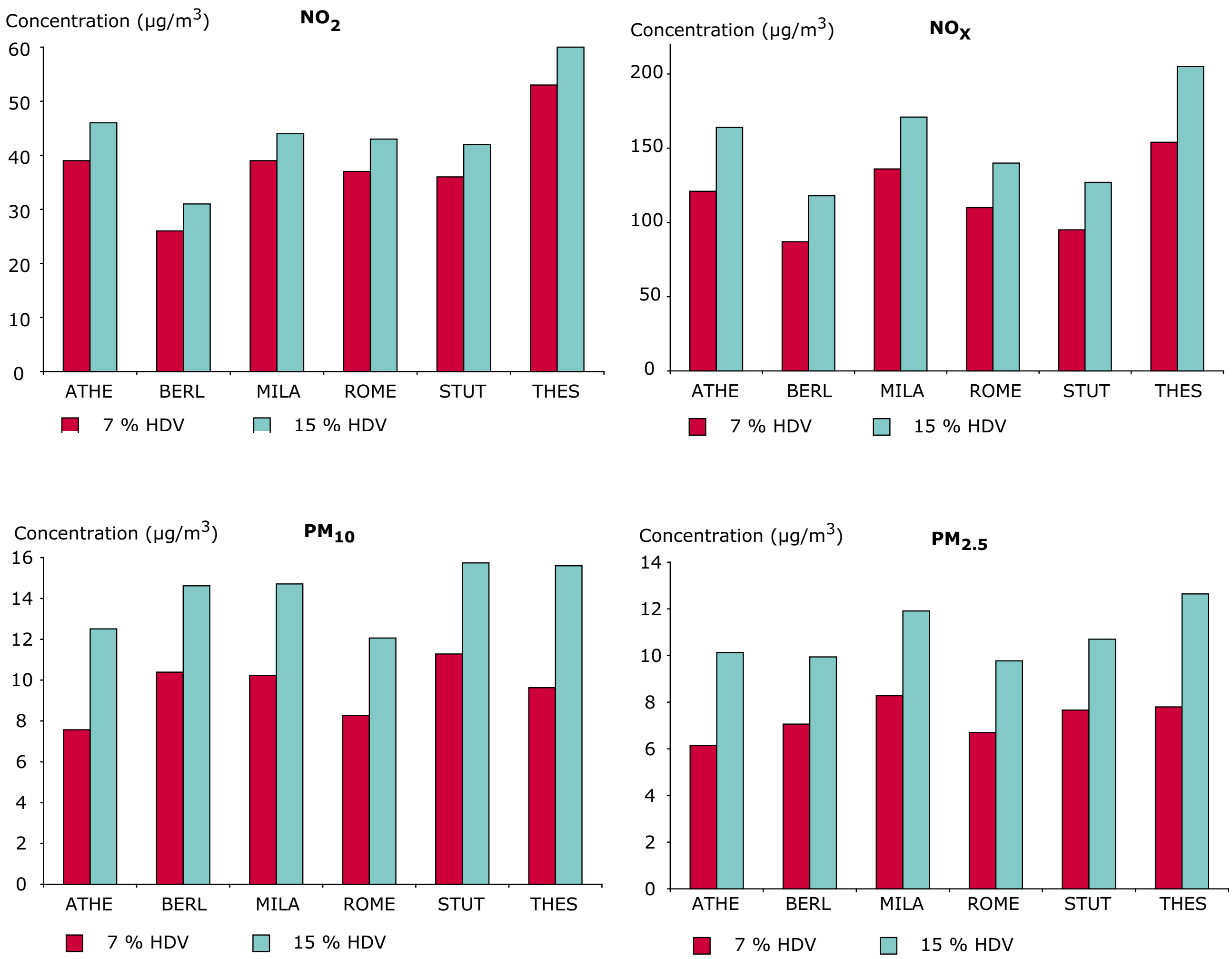
Figure 4.11 Number of daily PM_{10} exceedances of the 50 μg/m³ limit value in 20 European cities for the narrow canyon case



Note: The number of urban traffic stations available in each city is noted in brackets.

The consideration of a higher HDV % at street level increases all pollutant concentrations. However, this depends on the specific composition of the HDVs in each city. In countries such as Greece (Athens and Thessaloniki) where old technology and more polluting vehicles are still used, the increase is larger than in German or Italian cities. The NO_2 concentration increases by 5–7 µg/m³, NO_X by 30–51 µg/m³, PM_{10} by 4–6 µg/m³ and $PM_{2.5}$ by 3–5 µg/m³.

Figure 4.12 Mean annual NO_2, NO_X, PM_{10} and $PM_{2.5}$ street increments (µg/m³) in six European cities for a narrow street canyon with 20 000 vehicles per day, assuming a HDV % of 7 % and 15 %



4.1.2.2 *The influence of the different street canyon geometries*

In order to study the influence of the different canyon geometries on the street level concentrations, OSPM model results were computed for the three canyon types. Here, the same number of vehicles per day (20 000) was assumed. The results for PM_{10} are shown in Figures 4.13 and 4.14.

The highest street increments are observed in the narrow canyon case which due to its configuration has the effect of trapping the air pollutants inside the street. This results in high street level concentrations. Assuming the same amount of vehicles per day in the square and wide cases, the PM10 street increments are found to be lower by 33 % and 67 % compared to the concentrations in the narrow canyon.

Similar to the street increments, the largest number of exceedances is observed in the narrow canyon case. The model results show that for the reference year 2000, the allowed number of daily PM_{10} exceedances (35 days per year according to the 2005 limit value defined in Directive 1999/30/EC) is exceeded in almost all cities in the narrow canyon, in 14 cities in the square canyon and in half the cities in the wide canyon case.

Figure 4.13 Mean annual PM_{10} street increments (μg/m³) for the reference year 2000 in 20 European cities: model results for the narrow, square and wide canyons compared to observations

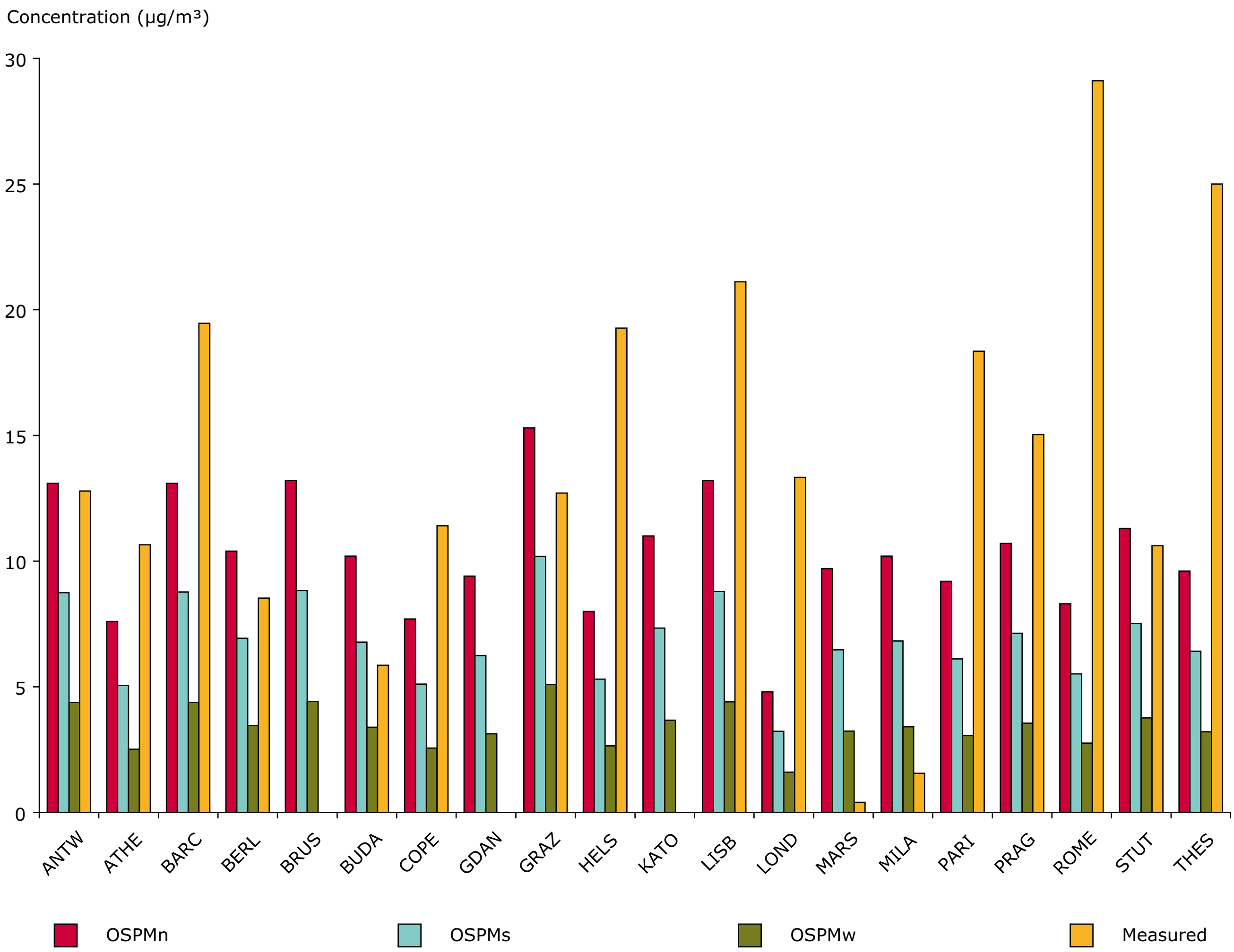
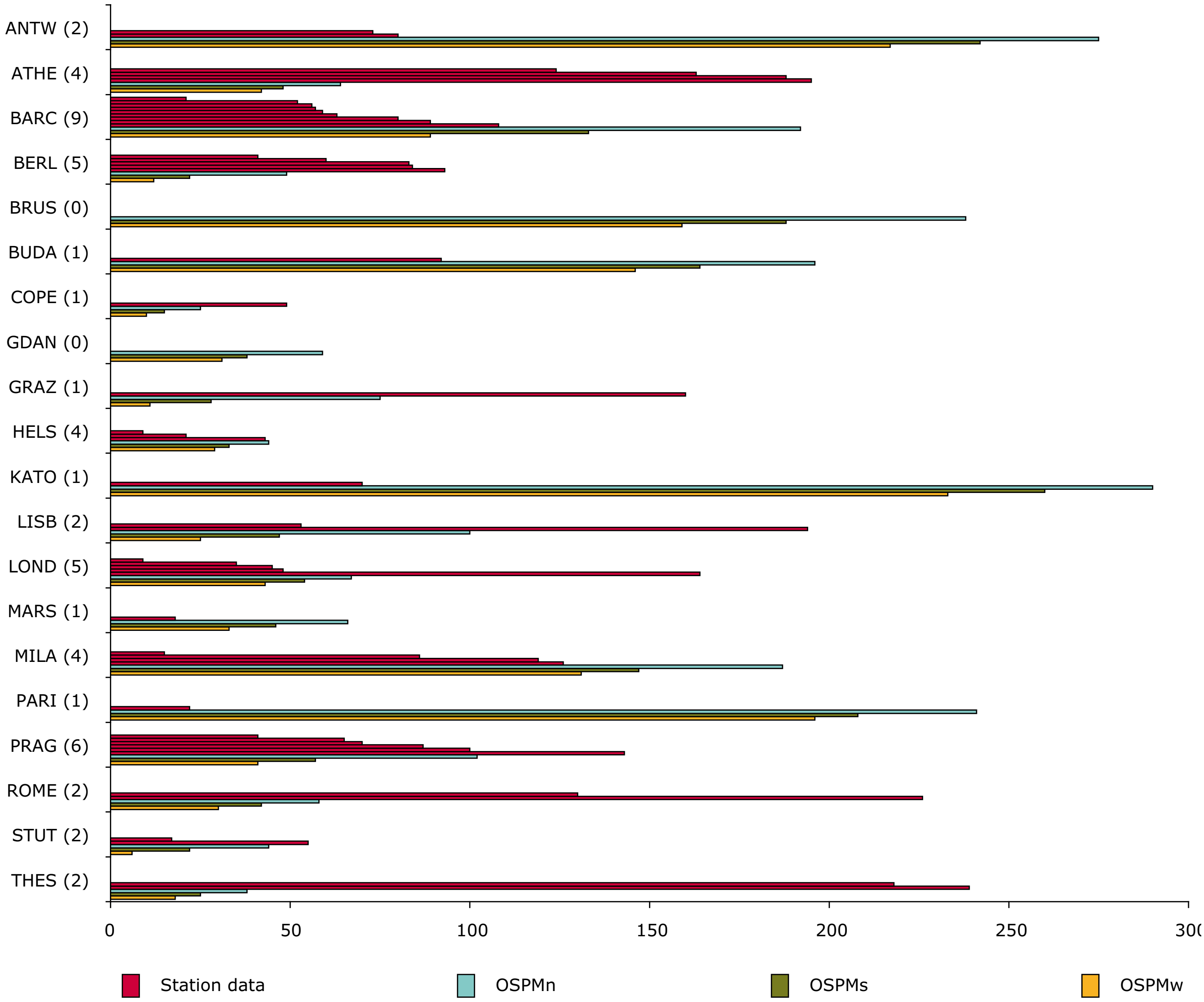


Figure 4.14 Number of daily PM_{10} exceedances of the 50 µg/m³ limit value in 20 European cities for the narrow, square and wide canyons for the reference year 2000



Note: The number of urban traffic stations available in each city is noted in brackets.

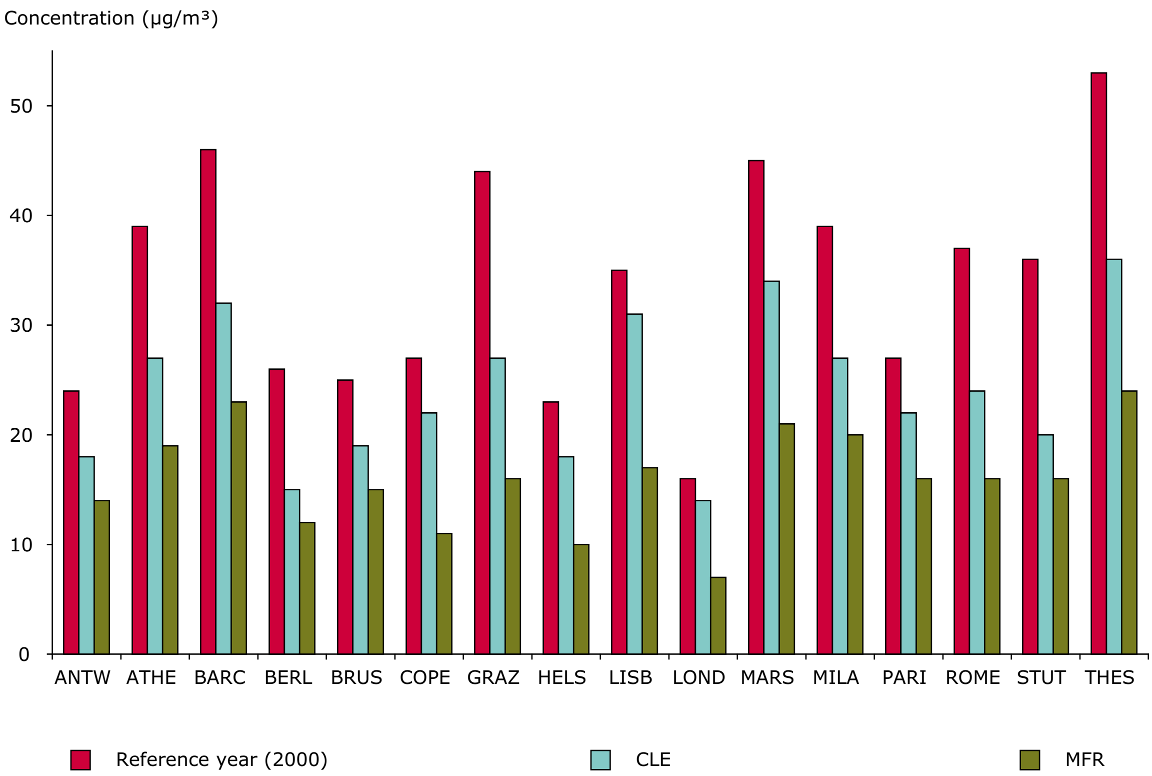
4.2 Scenarios

4.2.1 Street increments for 2030

In Figures 4.15 to 4.18, the street increments for the hypothetical narrow canyon (height 15 m, width 10 m and a traffic volume of 20 000 vehicles per day for the reference year 2000) are compared to the projected increments, according to the CLE and MFR scenarios (see Annex C and Cofala *et al.*, 2005 for details).

For the cities located in the non-EU-15 countries (Budapest, Gdansk, Katowice and Prague), the lack of reliable vehicle fleet data for 2000 results in a calculation of unrealistic attenuation factors for the projection year 2030. A reduction of around 95 % for both scenarios, both NO_x and PM was derived (see Annex C). As a result, the projected street increments for these countries were considered unrealistically low, unreliable and hence not included in the scenario analysis.

Figure 4.15 NO_2 annual mean street increments for cities across Europe in 2000 compared to the projected street increment in 2030



Note: The increments were calculated for the narrow canyon case using the CLE and MFR scenarios.

Figure 4.16 NO_x annual mean street increments for cities across Europe in 2000 compared to the projected street increment in 2030

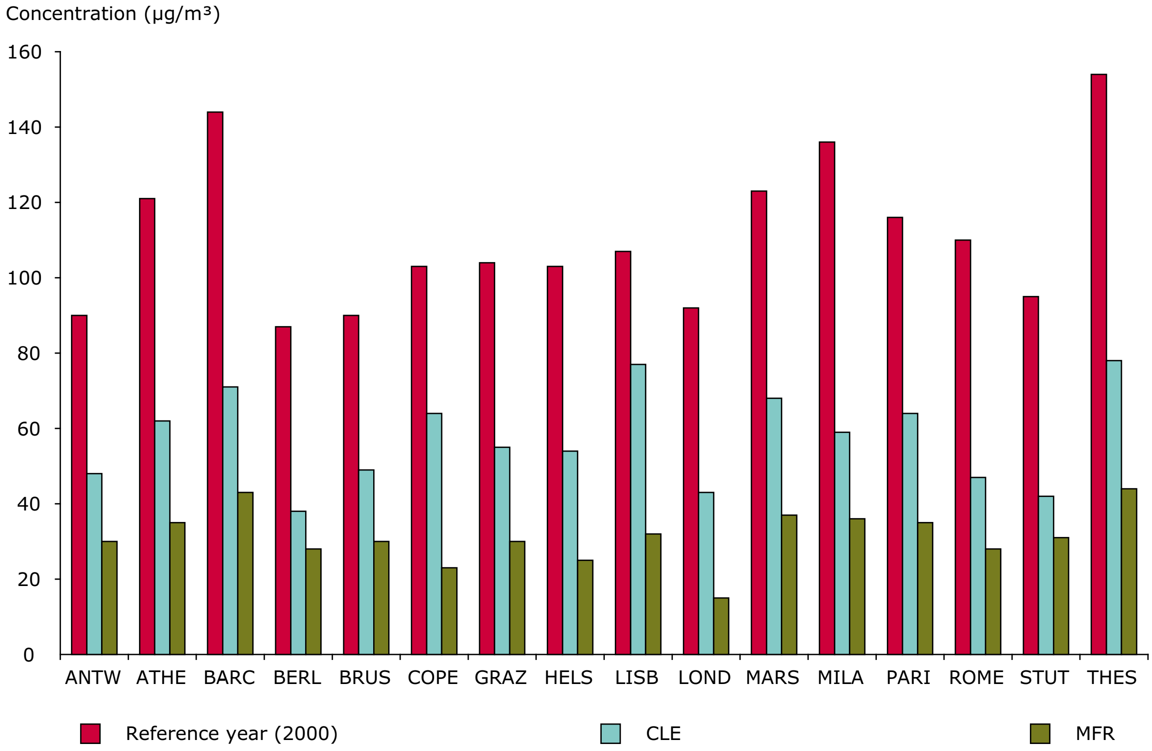
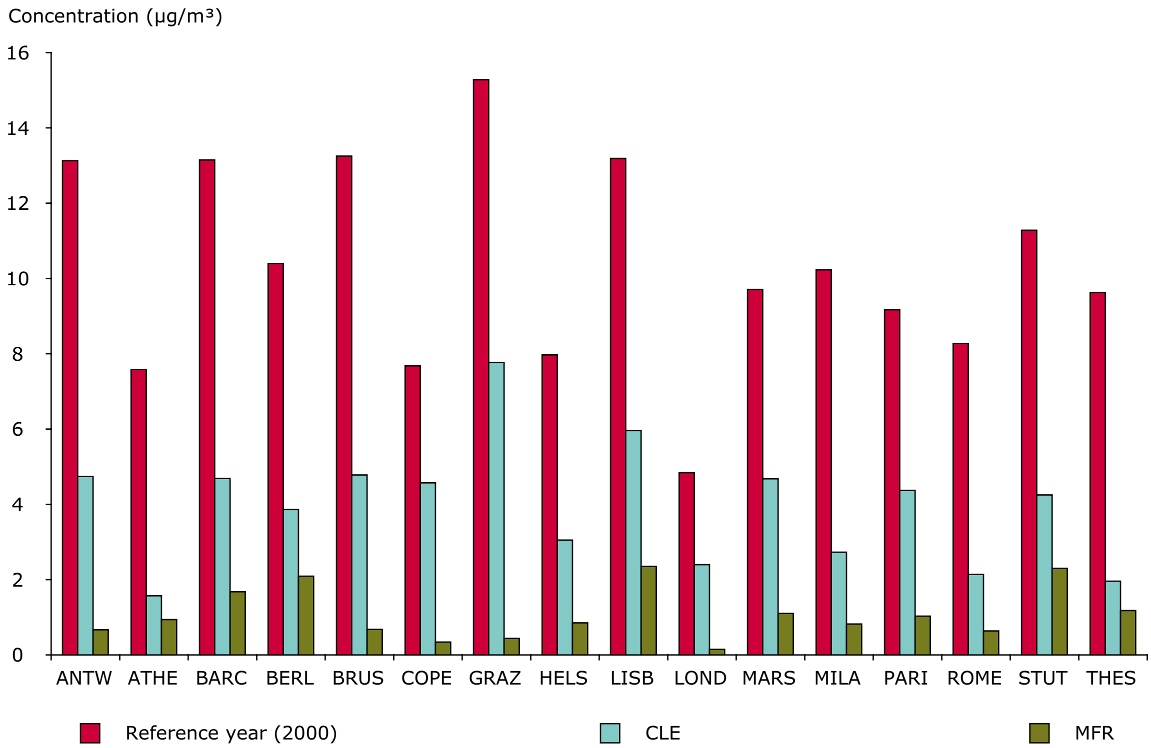


Figure 4.17 PM_{10} annual mean street increments for cities across Europe in 2000 compared to the projected street increment in 2030

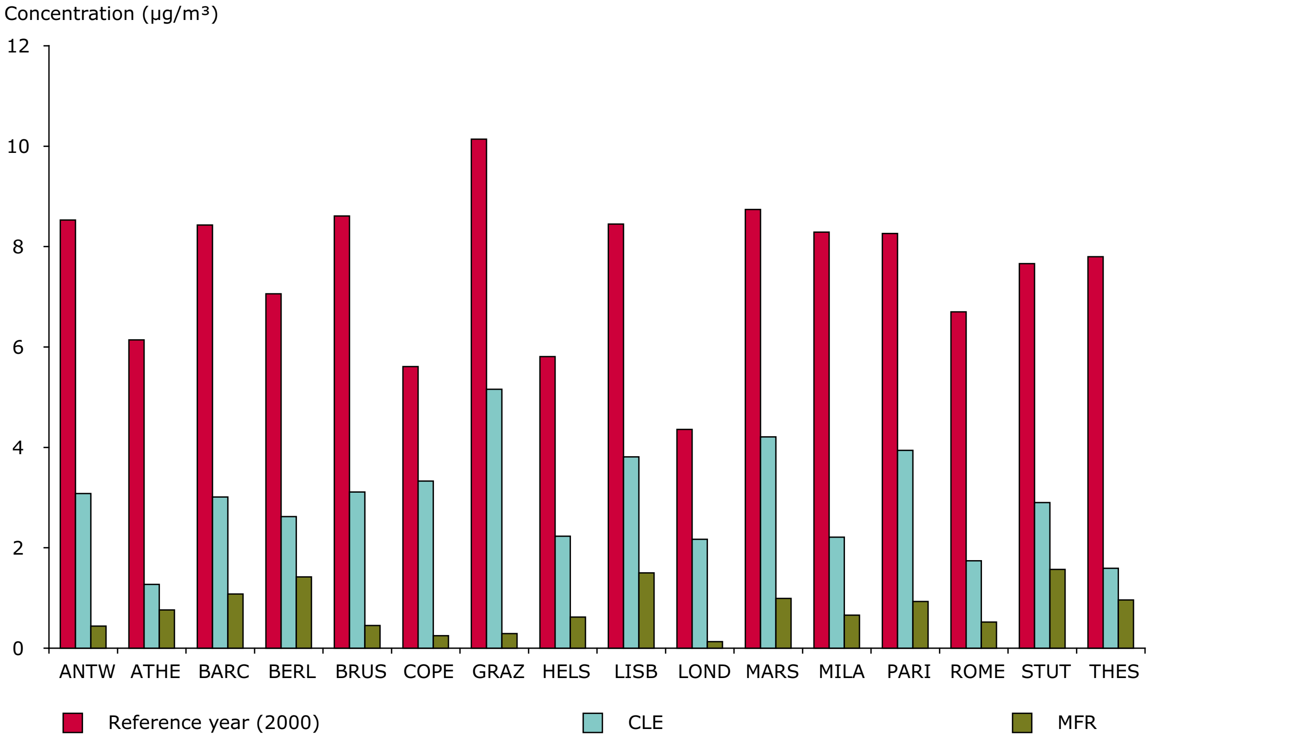


Note: The increments were calculated for the narrow canyon case using the CLE and MFR scenarios.

A reduced street increment is projected for all pollutants, according to both the CLE and MFR scenarios. For NO_2 the modelled street increment in 2000 ranged from 16–53 µg/m³ depending on the city. In 2030 it falls to 14–36 µg/m³ for the CLE and 7–24 µg/m³ for the MFR scenario. For NO_X the modelled street increment in 2000 ranged from 87–154 µg/m³, whereas in the CLE and MFR scenarios it is projected to range from 38–78 µg/m³ and 15–44 µg/m³ respectively. Larger reductions are projected for PM_{10} and $PM_{2.5}$. This range from 5–15 µg/m³ for PM_{10} in 2000 and a range of 2–8 µg/m³ and 0.2–2.4 µg/m³ is predicted for CLE and MFR respectively. For $PM_{2.5}$, the range of values from 4 to 10 µg/m³ in 2000 is projected to be between 1.3–5.2 µg/m³ and 0.1–1.6 µg/m³. These projections are in line with the significant reductions in the urban scale emissions, and hence the background concentrations and the street scale PM emissions attributed to the Euro V and Euro VI technology vehicles.

Regional air quality change is included in order to provide an impression of how important the street increment is in the reference year and also how its relative contribution changes in each scenario. In Figures 4.19 and 4.20 the change in the regional air quality (the city is located in an EMEP cell), the urban air quality (maximum OFIS results from Figures 4.1 and 4.3) and street scale air quality results (narrow canyon) are presented for a number of cities. As already mentioned, the maximum OFIS results for PM_{10} comprise a constant value of 17 µg/m³ (referred to as 'Natural' contribution in Figure 4.20). This accounts for PM sources such as windblown dust, sea salt and organic aerosols.

Figure 4.18 $PM_{2.5}$ annual mean street increments for cities across Europe in 2000 compared to the projected street increment in 2030



Note: The increments were calculated for the narrow canyon case using the CLE and MFR scenarios.

Figure 4.19 **NO_2 annual mean air quality at regional scale (EMEP), urban scale (OFIS) and street scale (OSPM) for cities across Europe in the reference year (2000) and the CLE and MFR scenarios**

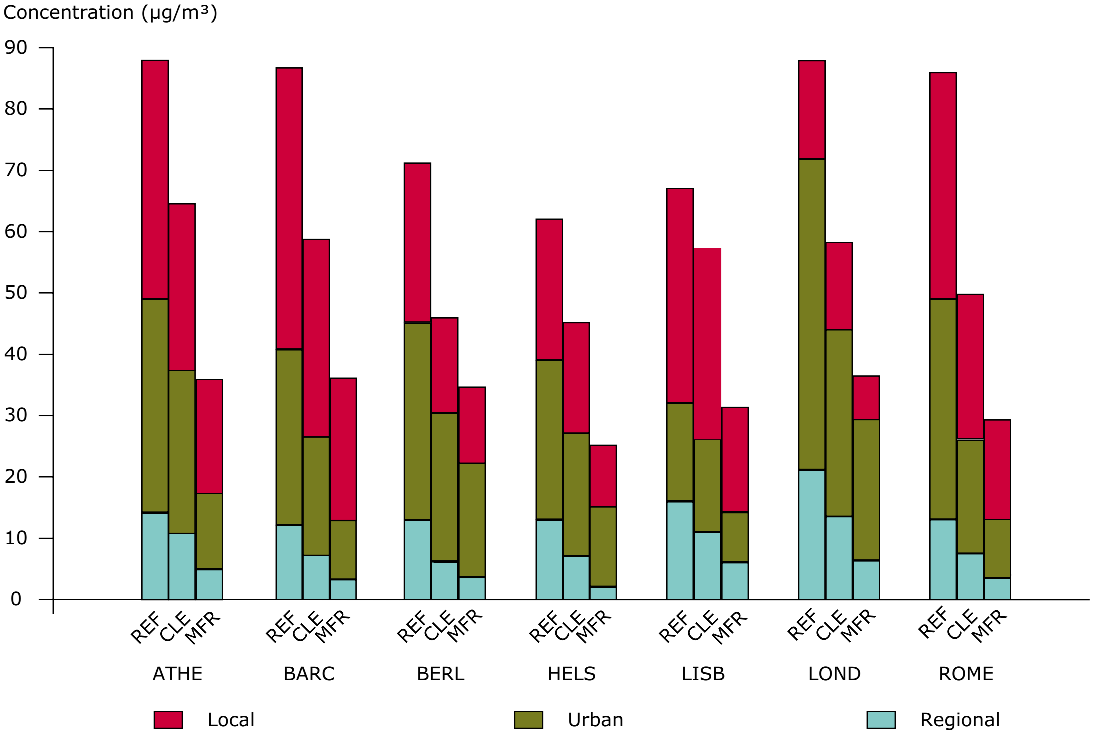
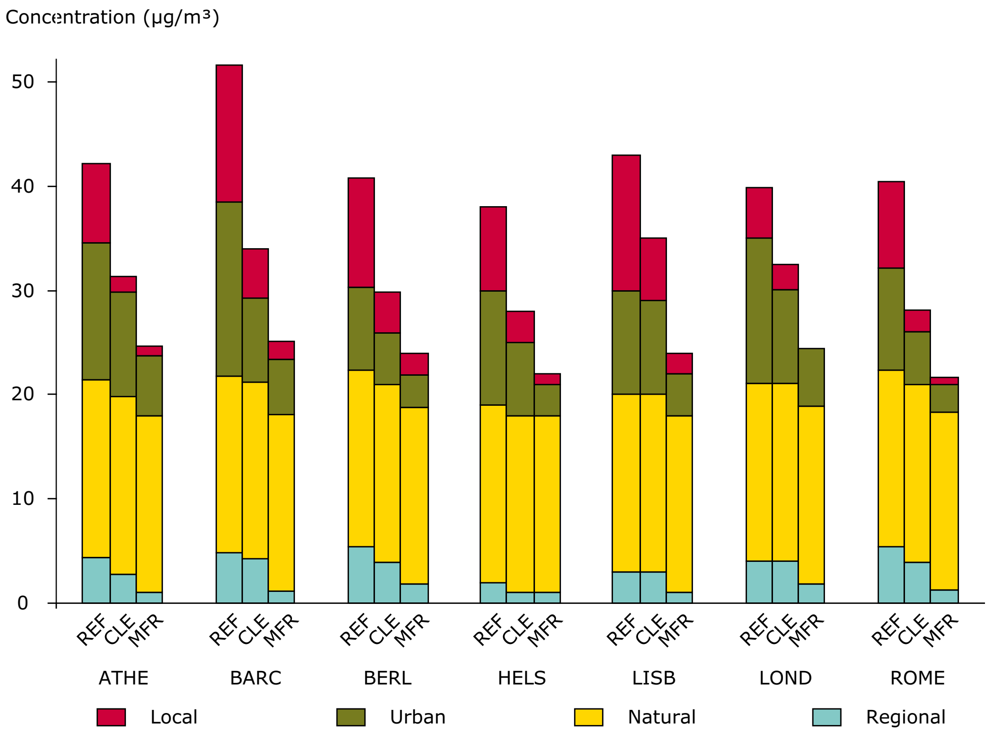


Figure 4.20 **PM_{10} annual mean air quality at regional scale (EMEP), urban scale (OFIS) and street scale (OSPM) for cities across Europe in the reference year (2000) and the CLE and MFR scenarios**



Note: The 'Natural' contribution refers to the constant value of 17 µg/m³, which accounts for PM sources such as windblown dust, sea salt and organic aerosols.

4.2.2 *Exceedance in 2030*

Figure 4.21 Number of hourly NO_2 exceedances of the 200 µg/m³ limit value in 20 European cities for the reference year 2000 and the CLE and MFR scenarios for 2030 calculated for the narrow canyon case

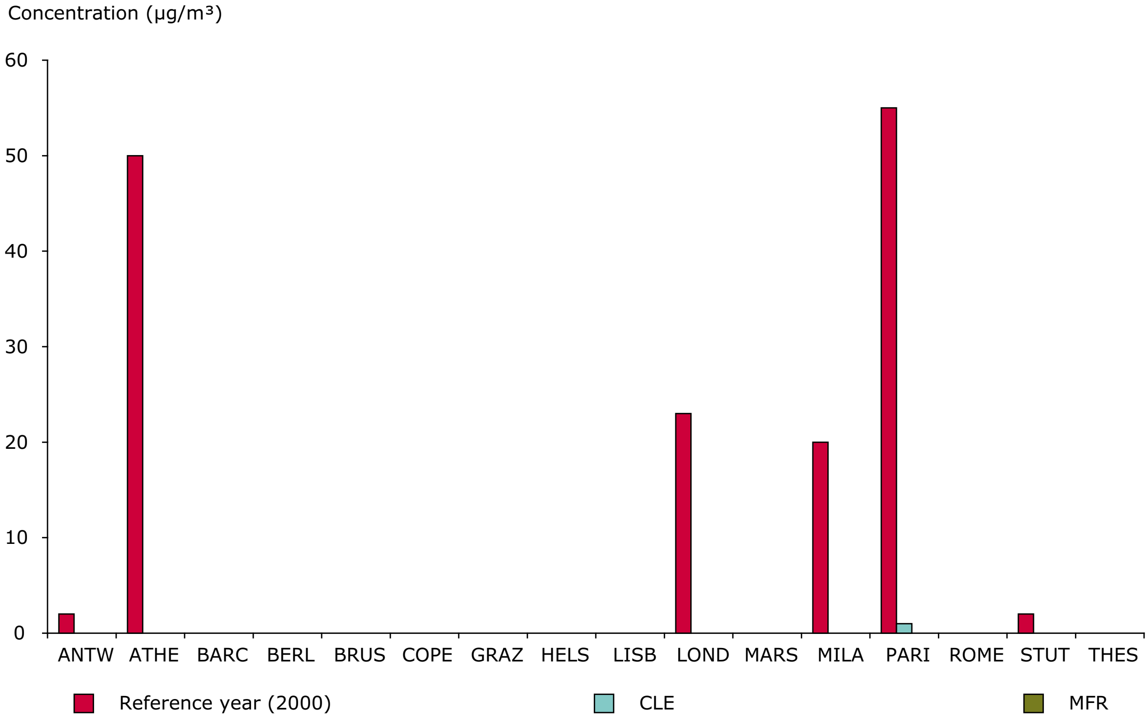
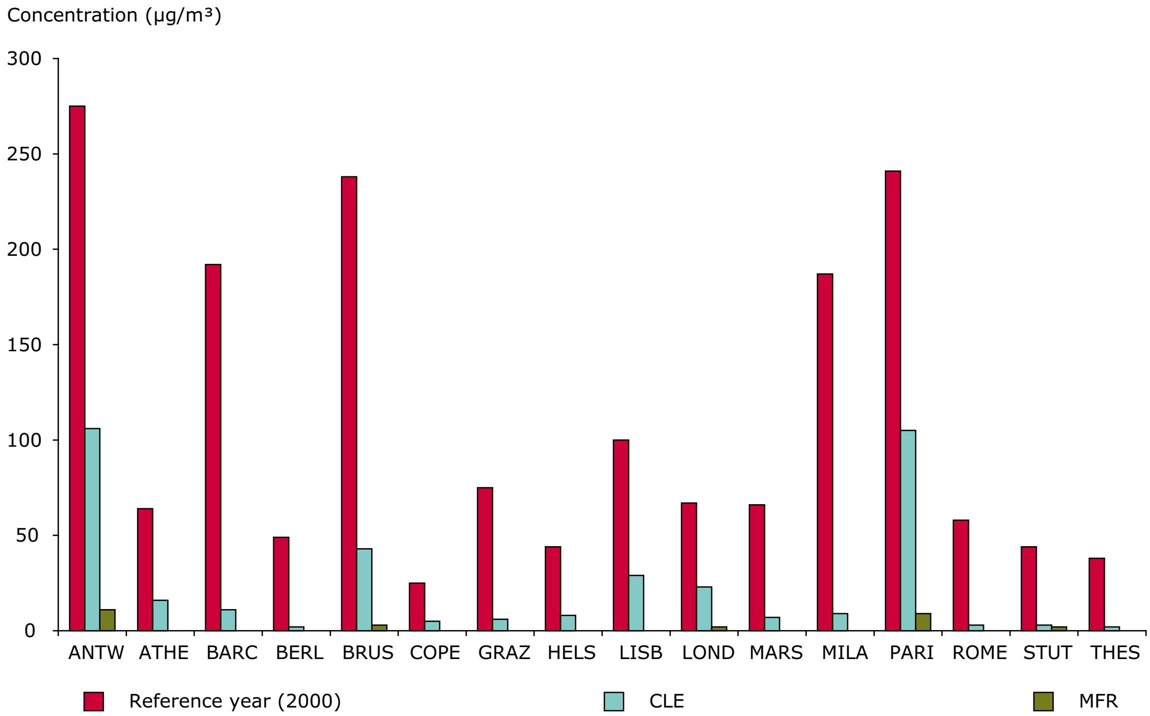


Figure 4.22 Number of daily PM_{10} exceedances of the 50 µg/m³ limit value in 20 European cities for the reference year 2000 and the CLE and MFR scenarios for 2030 calculated for the narrow canyon case



For all canyons, the number of exceedances for NO_2 and PM_{10} drops considerably in both the CLE and MFR scenarios. For the narrow canyon and for NO_2, almost no exceedances of the 2010 limit value (200 μg/m^3) are observed in 2030, according to the CLE scenario. Moreover, no exceedances are observed in the MFR scenario. For PM_{10}, no constant value has been added to account for missing natural PM_{10} emission sources. This lies in contrast to the approach followed in Section 4.1.1 (Figure 4.5). The reason for this difference is the uncertainty associated with the change of this value up until the projection year 2030.

Despite all limitations, Figure 4.22 provides useful information in terms of the relative change expected in the different cities, according to the two scenarios. The situation for PM_{10} is slightly different from that of NO_2. Although there is considerable reduction in the number of exceedances in the CLE scenario, the allowed number of exceedances (50 μg/m^3 not to be exceeded more than seven days a year, according to the 2010 indicative limit value) is still exceeded in nine cities. In the MFR scenario, all cities have close to zero exceedances except Antwerp and Paris, which are close but not below the allowed number of exceedances (11 and 9 days a year respectively). However, it should be noted that the worst street canyon cases have not been considered, and hence the allowed number of exceedances may still be exceeded (see Section 4.1.2). This is especially the case for PM_{10} where in most cases compliance is marginal. In view of the fact that the natural contribution to PM_{10} concentrations has not been considered in the scenario year 2030, it is highly likely that the 2010 PM_{10} limit value will be exceeded in 2030 in a number of cities.

5 Conclusions and future work

The complete regional-urban-local scale model cascade application used two models: OFIS and OSPM. The OFIS model (Arvanitis and Moussiopoulos, 2003) was driven by the regional-scale model EMEP (URL1). This was assumed to adequately describe the regional air quality around the city. The street-scale model OSPM (Berkowicz *et al.*, 1997) used OFIS model results to derive the urban background conditions required by the street scale model. This has proved successful in terms of analysing current and future air quality. The validation of model results against measurements has shown that the OFIS model can be used to adequately reproduce the urban background concentrations across the various cities. In terms of street level concentrations, the street canyon model OSPM has also successfully reproduced the street increments observed across cities.

The exceedances of the daily and hourly limit values calculated with OFIS and the comparison to measurements show that modelled exceedances compare well with measurements where the annual mean urban background concentrations estimates also compare well with the measured data. The successful comparison of urban scale model results and measured data depends critically on the appropriateness of the urban emission inventories assumed to represent city emissions. Small over-estimations or under-estimations of the model results compared to measurements in the annual mean concentrations result in larger differences in terms of exceedances. Due to the regional-urban-local scale modelling sequence followed in this approach, the accuracy of the model results for the urban background concentration significantly affects the street scale model output. The choice of the hypothetical street canyons will rarely coincide with the actual street canyon geometry and the specific traffic characteristics, which give the measured air quality data at traffic stations. The modelled air quality at street level will also be influenced by the urban background concentrations assumed, and thus by the urban emission inventories used. Finally, the accuracy of the street-scale model results for particular worst case hotspots would require extensive study of the worst-case street canyon configuration characteristics. This issue goes beyond the scope of the analysis.

Concerning the continuation of this type of work, there are various points to consider:

- The sensitivity of the street emissions and the consequent air quality calculations at local scale must be evaluated using the parameters of the Typology Methodology. In particular the vehicle speed, the street geometry, orientation and the HDV % must be further studied in terms of specific ranges of values and combinations of the various parameters. Here, the focus should be placed on worst case situations. Detailed local traffic measurements combined with air quality data for a range of cities and streets across Europe is required to support this work.

- The meteorological data used for the application of the street-scale model OSPM and derived from EMEP data must be compared as far as possible to actual measurements (roof-level meteorological data in each city). Model runs would need to support this comparison in terms of the impact of different wind speeds and wind directions on the street concentrations.

- The air quality projections for the non-EU-15 countries were considered unreliable since the emission attenuation factors initially calculated and used for the concentration estimates were based on unreliable vehicle fleet data for 2000 (see Annex C). The new emissions produced should be based on updated attenuation factors and used to assess the air quality also in the non-EU-15 countries.

- Due to the lack of data, it was assumed that the urban scale emission reductions were in line with the country scale emission reductions projected by the CLE and MFR scenarios for 2030. However, an estimate of the evolution of the city emissions according to specific local city development plans and urban population projections should be used instead of applying country level attenuation factors to the city level, as this could result in different projections of air quality in 2030 depending on the city growth rate and other factors.

- The EMEP model results for the CAFE scenarios were not available. Therefore, the projection year 2030 was chosen for the study instead. This is in line with 'The European Environment — State and outlook 2005' (EEA, 2005). Given that the air quality limit values have been set to apply from 2010 onwards, the air quality evolution according to the CAFE scenarios for 2010 and 2020 should also be studied.

- For the application of the urban scale model, detailed and gridded emission inventories must be made available. The MERLIN project prepared such gridded emission inventories using a top down approach (NUTS 3 down to the urban scale) through the application of the European Emission model (Friedrich and Reis, 2004). Such a top-down approach must be compared to bottom-up emission inventories based on local data in order to test the appropriateness of the spatial and temporal distributions assumed. A first step would be to compare the MERLIN emission inventories to local emission inventories (e.g. in the City-Delta project), where these are available. Depending on the findings and for certain cases, the air quality applications would need to be repeated.

- In line with the results of source apportionment studies across Europe, the contribution of natural PM sources may be re-evaluated.

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Annex A

SEC project layout

The work presented in this report is largely based on the findings of the Street Emission Ceilings (SEC) project of the ETC/ACC. This project aimed to develop a method for determining what local emission reductions in streets are needed to reach certain air quality thresholds. The work performed in 2003 and 2004 focused on four areas:

- analysis of concentration and traffic data from various station pairs (urban background and traffic station data);
- comparison with emission estimates using the COPERT 3 (Ntziachristos *et. al*, 2000) and TRENDS (Giannouli *et al.*, 2006) models;
- application of various street-scale models using the data available;
- development of a classification of street types, which at a later stage will allow a generalised approach for estimating the pollutant concentrations.

In order to study the excess concentrations observed at street-/roadside stations and test the specific models and tools against measurements, it was necessary to determine a number of cases where the data available would enable such a test basis. Moreover, it would then prove to be representative, and allow for generalisation of the results. Detailed quality controlled hourly traffic, meteorological data, street and urban background level concentrations (ideally $PM_{2.5}$, PM_{10}, NO_2, NO_X, CO and background O_3 were required), and appropriate street geometries were not readily available. Often, the exact location of the stations was a limiting factor for the analysis. Furthermore, the lack of detailed traffic data and incomplete datasets were also problems often encountered. Nevertheless, three case studies were singled out as most appropriate and for which an hourly data analysis for a full year was performed: Marylebone Road (London), Hornsgatan (Stockholm) and Frankfurter Allee (Berlin). The data analysis considered annual averages, monthly averages and average diurnal variations separately for weekdays/weekends and summer/winter periods. The analysis focused on street increments (the difference between street and urban background concentrations) and street increment ratios over NO_X. The differences in the street increments across the various cases reflected the differences in the average daily traffic, the type of canyon (open road or street canyon), the speed and heavy-duty vehicle fraction (henceforth HDV %) and the differences in the average wind speed. For PM_{10} the differences also reflected the use (or not) of studded tires. In the street increment ratios for NO_X, the average daily traffic, the street configuration and wind speed differences are in principle eliminated. The differences should reflect the variations in average emission factor ratios for the traffic flows due to differences in the HDV %. The results of this analysis allowed for an estimation of the strength of the road dust resuspension source to PM_{10} and $PM_{2.5}$. This was carried out by comparing street increment ratios over NO_X for winter/summer and workdays/weekends for PM_{10} and $PM_{2.5}$. For Marylebone Road (London), it was estimated that the resuspension source to PM_{10} is of about the same magnitude as the combined exhaust/brake/tyre wear source. In Hornsgatan (Stockholm) where studded tyres are used in winter, the resuspension source dominates PM_{10} relative to the exhaust. The resuspension source is significant even in the summer and it also gives a significant contribution to $PM_{2.5}$ in the street. Also in Frankfurter Allee (Berlin) the resuspension source is very strong and in relative terms larger than in Marylebone Road The results show that such an analysis can lead to a promising method of estimating 'emission factors' for the resuspension source, though more cases and more reliable data are needed in order to generalise the results (Larssen *et al.* 2004).

Comparison of the street increment ratios (PM/NO_X and CO/NO_X) with the corresponding emission ratios enabled site specific characteristics to emerge (e.g. importance of PM resuspension). It also provided a basis for the assessment of the air quality model applications that followed in terms of verifying the appropriateness of the emission factors for this type of work. The emission factor ratios compared well against the concentration ratios, and for PM the importance of non-tail pipe PM emissions (tyre and brake abrasion, road wear and dust resuspension) was particularly noted. Moreover, the comparison revealed that there is room for a significant resuspension source to $PM_{2.5}$ in Hornsgatan (Larssen *et al.* 2004).

Further to the concentration and emission ratio comparisons performed with data from individual sites, 'global' analysis was also performed using a number of stations across five European countries from Airbase. The comparison showed a fair agreement between the concentration and emission ratio of CO/NO_X at country level. This suggests that the measured concentrations originate from traffic-related emissions. The NO_X/PM and PM/CO emission ratios estimated by the COPERT 3 model were over- and underestimated respectively. This work highlighted once again the importance of PM emissions from gasoline-fuelled vehicles and non-exhaust sources which are not currently considered in the COPERT 3 model. Moreover, the lack of $PM_{2.5}$ concentration data on a broad scale across EU countries and for a number of stations was noted as a particular disadvantage for this type of comparison. Also, the fact that NO_X is not an 'obligatory' pollutant to be reported, according to the EoI Directive (only NO_2 must be reported), leads to additional data restrictions concerning both the concentrations analysis and the comparison with emission estimates (Mellios *et al.*, 2004).

The next step concerned the model applications. The data collected and studied for Marylebone Road (London), Hornsgatan (Stockholm) and Frankfurter Allee (Berlin), were further processed and the data sets were made available to interested institutes for performing a model intercomparison exercise (URL4). This exercise provided an insight into the level of uncertainty that is inherent in the various model calculations. It also supplied a first estimate of the uncertainty that enters from street level into a complete regional-urban-street scale model application. The large number of models that participated (13) and the variety of cases available enabled an evaluation of model performance, though it is important to bear in mind the restrictions of the input data. The model intercomparison exercise revealed that the models formulated specifically to describe pollutant dispersion in street canyons yield results closest to the actual measurements. In addition, easy-to-use models perform well and can be considered an appropriate tool for use by a non-expert user. OSPM results obtained by three different modelling groups were in agreement with each other, if one allows for the conclusion that user-introduced errors remain small for well-documented modelling tools. Overall, the semi-empirical models provided very satisfactory results and proved to be reliable for assessment purposes. The results for both the Frankfurter Allee and the Marylebone Road cases emphasise the importance of correct and representative input data, and the need for a consequent sensitivity analysis. The scientific community verified its interest in participating in such exercises provided that complete and reliable datasets are made available. The insufficient number of representative datasets was noted as a particular problem in conducting such model intercomparison exercises (Moussiopoulos *et al.*, 2004).

In parallel with the above data analysis and modelling activities, the theoretical basis for the classification of street types ('street typology') was developed. This typology would allow for a generalised methodology to determine the local emission reductions needed to reach certain air quality thresholds. In the development of the typology methodology, the balance had to be maintained between model accuracy, which requires many explicit and continuous parameters, and simplicity, which demands giving preference to classified parameters. A first selection of the key parameters sufficiently characterising the various street classes resulted in the distinction of twelve street types. The classified parameters (represented by ranges of values) consisted of geometry (street canyon or not), HVD %, traffic behaviour (speed) and distance of the receptor from the road axis. The only parameter retained as explicit and continuous was daily traffic intensity. The candidate parameters were assessed in terms of their importance to air pollution, their suitability for air quality modelling and the availability of data (on specific streets and statistics across Europe). A further criterion was whether the particular parameter could be altered by specific measures. For example, the HDV % is important since it is a vehicle category with significant air emissions, but technological improvements related to emission reduction for HDVs and private cars follow different tracks in time. In the further development of the typology methodology, an iterative procedure is envisaged using various models. Here, a sensitivity analysis is performed in terms of the parameters and the values selected. After the model applications take place, the typology may be improved (van den Hout and Teeuwisse, 2004).

Overall, the main problem noted throughout SEC and the various applications was the lack of complete and reliable datasets. In terms of street model applications, the urban background concentrations, the meteorological data and the lack of specific street canyon traffic data were noted as particularly limiting factors. In order to assess street level concentrations across a number of cases, the well documented semi-empirical street canyon model OSPM was selected and applied. This also provided good results in the model intercomparison exercise. The background concentrations were assumed to be adequately described by the urban scale model OFIS, which was in turn driven by the regional scale model EMEP. This was assumed to adequately describe regional air quality. The complete regional-urban-local scale modelling sequence was applied for the case study from Berlin. Here, a full measurement dataset was also available. Model results compared well both against measurements and against the application using the full measurement dataset. The importance of the successful application of such a modelling sequence is obvious since it allows for the assessment of future air quality considering policies and measures affecting the regional, urban and local scale.

Annex B

Table B1 Monitoring stations used for comparisons between observations and model result

	Station name	Station type	Type of area	NO_2		NO_x		PM_{10}		$PM_{2.5}$	
				year	data.av.	year	data.av.	year	data.av.	year	data.av.
ANTWERP											
BE0227A	42R821:Beveren	Background	Suburban	2003	95 %	2000	97 %	2003	*	2003	*
BE0232A	42R811:Schoten	Background	Suburban	2003	70 %	2000	*	2003	*	2003	*
BE0432A	40TR01:Temse	Background	Suburban	2003	*	2000	*	2003	100 %	2003	*
BE0204A	42M802:Antwerpen	Traffic	Urban	2003	*	2000	*	2003	74 %	2003	*
BE0228A	42R801:Borgerhout	Traffic	Urban	2003	89 %	2000	93 %	2003	94 %	2003	76 %
ATHENS											
GR0027A	Liosia	Background	Suburban	2001	99 %	2001	99 %	2001	*	*	*
GR0037A	Thrakomakedones	Background	Suburban	2001	94 %	2001	95 %	2001	99 %	*	*
GR0039A	Agia Paraskevi	Background	Suburban	2001	82 %	2001	84 %	2001	73 %	*	*
GR0043A	Zografou	Background	Suburban	2001	98 %	2001	98 %	2001	97 %	*	*
GR0028A	Peristeri	Background	Urban	2001	89 %	2001	89 %	2001	*	*	*
GR0031A	Nea Smirni	Background	Urban	2001	91 %	2001	92 %	2001	*	*	*
GR0041A	Pireaus-2	Background	Urban	2001	99 %	2001	99 %	2001	*	*	*
GR0042A	Galatsi	Background	Urban	2001	86 %	2001	85 %	2001	*	*	*
GR0002A	Athinas	Traffic	Urban	2001	92 %	2001	92 %	2001	*	*	*
GR0003A	Aristotelous	Traffic	Urban	2001	91 %	2001	*	2001	96 %	*	*
GR0022A	Marousi	Traffic	Urban	2001	99 %	2001	99 %	2001	98 %	*	*
GR0030A	Pireaus-1	Traffic	Urban	2001	86 %	2001	88 %	2001	52 %	*	*
GR0032A	Patision	Traffic	Urban	2001	96 %	2001	96 %	2001	*	*	*
GR0040A	Goudi	Traffic	Urban	2001	97 %	2001	97 %	2001	94 %	*	*
BARCELONA											
ES1024A	ES1024A:Mirador	Background	Urban	2001	*	2001	*	2003	15 %	*	*
ES0559A	ES0559A:Plaηa Universitat	Traffic	Urban	2001	*	2001	*	2003	19 %	*	*
ES0691A	ES0691A-I2:Poble Nou	Traffic	Urban	2001	84 %	2001	84 %	2003	*	*	*
ES0692A	ES0692A-I3:L'Hospitalet	Traffic	Urban	2001	93 %	2001	93 %	2003	34 %	*	*
ES0693A	ES0693A-I5:Badalona	Traffic	Urban	2001	52 %	2001	52 %	2003	*	*	*
ES1018A	ES1018A-E1:Terrassa	Traffic	Urban	2001	89 %	2001	89 %	2003	88 %	*	*
ES1231A	ES1231A-AT:Sant Cugat del Valles	Traffic	Urban	2001	88 %	2001	87 %	2003	25 %	*	*
ES1262A	ES1262A:Ad-Sabadell	Traffic	Urban	2001	78 %	2001	78 %	2003	28 %	*	*
ES1362A	ES1362A:Eugeni D'Ors	Traffic	Urban	2001	*	2001	*	2003	36 %	*	*
ES1396A	ES1396A:ID-Barcelona	Traffic	Urban	2001	96 %	2001	96 %	2003	36 %	*	*
ES1438A	ES1438A:IH-Barcelona(example)	Traffic	Urban	2001	67 %	2001	65 %	2003	36 %	*	*
ES1453A	ES1453A-II:Torreballdovina	Traffic	Urban	2001	95 %	2001	95 %	2003	*	*	*
ES1480A	ES1480A-IJ-Gracia-Sant Gervasi	Traffic	Urban	2001	78 %	2001	78 %	2003	38 %	*	*
ES1551A	ES1551A-B9-Barberα del Vallıs	Traffic	Urban	2001	89 %	2001	89 %	2003	*	*	*
BERLIN											
DE1091A	DEBE051:B Buch	Background	Suburban	2000	98 %	2000	98 %	2002	98 %	2003	*
DE1101A	DEBB031:Konigs Wusterhausen	Background	Suburban	2000	93 %	2000	*	2002	98 %	2003	*
DE1210A	DEBB050:Bernau	Background	Suburban	2000	99 %	2000	*	2002	*	2003	*
DE1212A	DEBB052:Potsdam-Michendorfer Chaussee	Background	Suburban	2000	*	2000	*	2002	95 %	2003	*
DE0742A	DEBE034:B Neukolln-Nansenstraie	Background	Urban	2000	98 %	2000	98 %	2002	98 %	2003	*
DE0982A	DEBB021:Potsdam-Zentrum	Background	Urban	2000	83 %	2000	*	2002	99 %	2003	98 %

Table B1 Monitoring stations used for comparisons between observations and model result, cont.

	Station name	Station type	Type of area	NO_2 year	NO_2 data.av.	NO_X year	NO_X data.av.	PM_{10} year	PM_{10} data.av.	$PM_{2.5}$ year	$PM_{2.5}$ data.av.
DE1227A	DEBE066:B Karlshorst-Rheingoldstr./Konigswinterstr.	Background	Urban	2000	95 %	2000	95 %	2002	*	2003	*
DE0715A	DEBE010:B Wedding-Amrumer Str.	Traffic	Urban	2000	96 %	2000	96 %	2002	98 %	2003	*
DE0773A	DEBE014:B Charlottenburg-Stadtautobahn	Traffic	Urban	2000	96 %	2000	96 %	2002	97 %	2003	*
DE0946A	DEBE044:B Mitte-Parochialstr.	Traffic	Urban	2000	95 %	2000	*	2002	94 %	2003	*
DE1111A	DEBE064:B Neukolln-Karl-Marx-Str. 76	Traffic	Urban	2000	99 %	2000	99 %	2002	*	2003	*
DE1115A	DEBE065:B Friedrichshain-Frankfurter Allee	Traffic	Urban	2000	98 %	2000	98 %	2002	94 %	2003	*
DE1169A	DEBE061:B Steglitz-Schildhornstr.	Traffic	Urban	2000	98 %	2000	98 %	2002	98 %	2003	*
DE1188A	DEBE063:B Neukolln-Silbersteinstr.	Traffic	Urban	2000	91 %	2000	91 %	2002	*	2003	*
BRUSSELS											
BE0185A	41N043:Haren	Background	Suburban	2000	90 %	2000	89 %	2000	95 %	2000	*
BE0186A	41R012:UCCLE	Background	Suburban	2000	95 %	2000	93 %	2000	99 %	2000	*
BE0192A	42R010:St.sStevensw	Background	Suburban	2000	*	2000	*	2000	*	2000	*
BE0309A	41B011:Berchem S.A	Background	Suburban	2000	95 %	2000	93 %	2000	99 %	2000	*
BE0371A	41MEU1:Meudon	Background	Suburban	2000	88 %	2000	86 %	2000	99 %	2000	97 %
BE0422A	40SZ01:Steenokkerz	Background	Suburban	2000	*	2000	*	2000	*	2000	*
BE0423A	40SZ02:Steenokkerz	Background	Suburban	2000	*	2000	*	2000	*	2000	*
BE0184A	41R001:Molenbeek	Background	Urban	2000	92 %	2000	91 %	2000	99 %	2000	50 %
BE0395A	41B004:Ste.Catheri	Background	Urban	2000	*	2000	*	2000	*	2000	*
BE0403A	41B006:Parl.Europe	Background	Urban	2000	*	2000	*	2000	*	2000	*
BE0308A	41B003:Arts-Loi	Traffic	Urban	2000	95 %	2000	93 %	2000	*	2000	*
BE0402A	41B005:Belliard	Traffic	Urban	2000	*	2000	*	2000	*	2000	*
BUDAPEST											
HU0022A	Budapest Gilice tır	Background	Suburban	2003	93 %	2003	93 %	2003	63 %	*	*
HU0032A	Szαzhalombatta	Background	Suburban	2003	*	2003	*	2003	*	*	*
HU0021A	Budapest Baross tır	Traffic	Urban	2003	46 %	2003	47 %	2003	39 %	*	*
COPENHAGEN											
DK0045A	Copenhagen/1259	Background	Urban	2002	90 %	2000	97 %	2002	61 %	*	*
DK0030A	Copenhagen/1257	Traffic	Urban	2002	99 %	2000	98 %	2002	89 %	*	*
DK0034A	H.C.Andersens Boulevard, City	Traffic	Urban	2002	98 %	2000	*	2002	*	*	*
GDANSK											
PL0045A	GdanskPW1	Background	Urban	2000	99 %	2000	99 %	2000	57 %	*	*
PL0046A	GdanskKa2	Background	Urban	2000	93 %	2000	94 %	2000	93 %	*	*
PL0047A	GdanskWy3	Background	Urban	2000	99 %	2000	98 %	2000	72 %	*	*
PL0049A	GdanskOs5	Background	Urban	2000	*	2000	*	2000	65 %	*	*
PL0050A	SopotBP6	Background	Urban	2000	100 %	2000	100 %	2000	100 %	*	*
PL0052A	GdanskLe8	Background	Urban	2000	97 %	2000	98 %	2000	99 %	*	*
GRAZ											
AT0022A	Graz Nord	Background	Suburban	2000	95 %	2000	95 %	2001	89 %	*	*
AT0085A	Graz Sòd	Background	Suburban	2000	99 %	2000	*	2001	*	*	*
AT0087A	Graz West	Background	Suburban	2000	96 %	2000	96 %	2001	*	*	*
AT0112A	Graz Ost	Background	Suburban	2000	99 %	2000	99 %	2001	76 %	*	*
AT0119A	Graz Platte	Background	Suburban	2000	*	2000	*	2001	*	*	*
AT0109A	Graz Mitte	Background	Urban	2000	89 %	2000	89 %	2001	69 %	*	*
AT0118A	Graz Schlossberg	Background	Urban	2000	*	2000	*	2001	*	*	*
AT0217A	Graz Tiergartenweg	Background	Urban	2000	*	2000	*	2001	*	*	*
AT0205A	Graz Don Bosco	Traffic	Urban	2000	88 %	2000	88 %	2001	94 %	*	*

Table B1 Monitoring stations used for comparisons between observations and model result, cont.

	Station name	Station type	Type of area	NO_2		NO_x		PM_{10}		$PM_{2.5}$	
				year	data.av.	year	data.av.	year	data.av.	year	data.av.
HELSINKI									T-B		
FI0050A	Tikkurila 2	Background	Urban	2003	*	2000	*	2003	*	2003	*
FI0124A	Kallio 2	Background	Urban	2003	100 %	2000	99 %	2003	98 %	2003	95 %
FI0004A	Vallila 1	Traffic	Urban	2003	99 %	2000	96 %	2003	98 %	2003	96 %
FI0006A	Vallila 2	Traffic	Urban	2003	*	2000	*	2003	45 %	2003	*
FI0018A	Töölö	Traffic	Urban	2003	100 %	2000	99 %	2003	99 %	2003	*
FI0142A	Runeberginkatu	Traffic	Urban	2003	84 %	2000	*	2003	84 %	2003	84 %
KATOWICE											
PL0008A	KatowZal	Background	Urban	2000	98 %	2000	72 %	2000	*	*	*
PL0022A	KatowRac	Background	Urban	2000	92 %	2000	92 %	2000	99 %	*	*
PL0040A	Chorzow	Background	Urban	2000	96 %	2000	96 %	2000	73 %	*	*
PL0042A	PiekarySl	Background	Urban	2000	96 %	2000	96 %	2000	97 %	*	*
PL0043A	Wojkowice	Background	Urban	2000	95 %	2000	95 %	2000	99 %	*	*
PL0041A	Sosnowiec	Traffic	Urban	2000	97 %	2000	97 %	2000	96 %	*	*
LISBON											
PT0087A	Olivais	Background	Urban	2001	95 %	2000	93 %	2001	95 %	2002	*
PT0090A	Chelas	Background	Urban	2001	98 %	2000	97 %	2001	*	2002	*
PT0091A	Beato	Background	Urban	2001	99 %	2000	91 %	2001	*	2002	*
PT0106A	Paio Pires aut.	Background	Urban	2001	95 %	2000	54 %	2001	*	2002	*
PT0109A	Alfragide/Amadora	Background	Urban	2001	97 %	2000	*	2001	*	2002	51 %
PT0110A	Laranjeiro	Background	Urban	2001	93 %	2000	*	2001	81 %	2002	*
PT0111A	Reboleira	Background	Urban	2001	75 %	2000	*	2001	81 %	2002	*
PT0112A	Loures	Background	Urban	2001	*	2000	*	2001	58 %	2002	*
PT0114A	Escavadeira II	Background	Urban	2001	*	2000	*	2001	*	2002	*
PT0115A	Restelo	Background	Urban	2001	*	2000	*	2001	*	2002	*
PT0059A	Hospital Velho	Traffic	Urban	2001	81 %	2000	92 %	2001	*	2002	*
PT0088A	Entrecampos	Traffic	Urban	2001	83 %	2000	89 %	2001	78 %	2002	80 %
PT0089A	Benfica	Traffic	Urban	2001	95 %	2000	92 %	2001	*	2002	*
PT0093A	Avenida da Liberdade	Traffic	Urban	2001	95 %	2000	88 %	2001	98 %	2002	*
PT0108A	Câmara Municipal	Traffic	Urban	2001	96 %	2000	60 %	2001	*	2002	*
LONDON											
GB0586A	London Eltham	Background	Suburban	2000	97 %	2000	97 %	2003	99 %	2000	*
GB0608A	London Bexley	Background	Suburban	2000	97 %	2000	96 %	2003	96 %	2000	*
GB0621A	London Sutton (sut3)	Background	Suburban	2000	91 %	2000	91 %	2003	*	2000	*
GB0642A	London Hillingdon	Background	Suburban	2000	98 %	2000	97 %	2003	87 %	2000	*
GB0420A	West London	Background	Urban	2000	98 %	2000	97 %	2003	*	2000	*
GB0566A	London Bloomsbury	Background	Urban	2000	96 %	2000	95 %	2003	58 %	2000	94 %
GB0616A	London Brent	Background	Urban	2000	98 %	2000	97 %	2003	94 %	2000	*
GB0620A	London N.Kensington	Background	Urban	2000	96 %	2000	96 %	2003	98 %	2000	*
GB0622A	London Wandsworth	Background	Urban	2000	97 %	2000	97 %	2003	*	2000	*
GB0638A	London Haringey	Background	Urban	2000	*	2000	*	2003	*	2000	*
GB0644A	London Teddington	Background	Urban	2000	99 %	2000	98 %	2003	*	2000	*
GB0645A	Thurrock	Background	Urban	2000	93 %	2000	92 %	2003	98 %	2000	*
GB0650A	London Hackney	Background	Urban	2000	91 %	2000	91 %	2003	*	2000	*
GB0656A	London Southwark	Background	Urban	2000	96 %	2000	96 %	2003	*	2000	*
GB0672A	London Lewisham	Background	Urban	2000	43 %	2000	43 %	2003	*	2000	*
GB0743A	London Westminster	Background	Urban	2000	*	2000	*	2003	64 %	2000	*
GB0623A	Sutton Roadside (sut1)	Traffic	Urban	2000	87 %	2000	87 %	2003	*	2000	*
GB0624A	Tower Hamlets Roadside	Traffic	Urban	2000	91 %	2000	90 %	2003	*	2000	*
GB0636A	Camden Kerbside	Traffic	Urban	2000	96 %	2000	96 %	2003	99 %	2000	*

Table B1 Monitoring stations used for comparisons between observations and model result, cont.

	Station name	Station type	Type of area	NO_2		NO_x		PM_{10}		$PM_{2.5}$	
				year	data.av.	year	data.av.	year	data.av.	year	data.av.
GB0637A	Haringey Roadside	Traffic	Urban	2000	88 %	2000	88 %	2003	98 %	2000	*
GB0659A	London A3 Roadside	Traffic	Urban	2000	97 %	2000	96 %	2003	95 %	2000	*
GB0667A	Southwark Roadside	Traffic	Urban	2000	90 %	2000	89 %	2003	*	2000	*
GB0682A	London Marylebone Road	Traffic	Urban	2000	96 %	2000	96 %	2003	99 %	2000	99 %
GB0685A	Hounslow Roadside (HS1)	Traffic	Urban	2000	97 %	2000	97 %	2003	*	2000	*
GB0695A	London Cromwell Road 2	Traffic	Urban	2000	94 %	2000	93 %	2003	*	2000	*
GB0697A	London Bromley	Traffic	Urban	2000	82 %	2000	83 %	2003	*	2000	*
GB0774A	Brentford Roadside	Traffic	Urban	2000	*	2000	*	2003	*	2000	*
MARSEILLES											
FR1108A	Saint Louis	Background	Suburban	2000	96 %	*	*	2002	97 %	2002	95 %
FR1109A	Aubagne Penitents	Background	Suburban	2000	95 %	*	*	2002	*	2002	*
FR1112A	Plan de Cuques	Background	Suburban	2000	98 %	*	*	2002	*	2002	*
FR1114A	P/Huveaune Gymnase	Background	Suburban	2000	96 %	*	*	2002	*	2002	*
FR1116A	Cinq Avenues	Background	Suburban	2000	97 %	*	*	2002	95 %	2002	*
FR1117A	Ste Marguerite	Background	Suburban	2000	94 %	*	*	2002	*	2002	*
FR1115A	Marseille Prado	Background	Urban	2000	97 %	*	*	2002	*	2002	*
FR1119A	Marseille Thiers Noa	Background	Urban	2000	96 %	*	*	2002	95 %	2002	*
FR0177A	Timone	Traffic	Urban	2000	97 %	*	*	2002	96 %	2002	*
MILAN											
IT1017A	P.CO Lambro 301530	Background	Suburban	2000	99 %	2000	99 %	2003	*	*	*
IT0466A	Juvara 301518	Background	Urban	2000	98 %	2000	98 %	2003	98 %	*	*
IT0706A	Limito 301524	Background	Urban	2000	*	2000	*	2003	88 %	*	*
IT1020A	Via Messina 301541	Background	Urban	2000	*	2000	*	2003	*	*	*
IT1034A	Meda 301527	Background	Urban	2000	98 %	2000	98 %	2003	96 %	*	*
IT0467A	Zavattari 301544	Traffic	Urban	2000	97 %	2000	96 %	2003	*	*	*
IT0477A	Marche 301526	Traffic	Urban	2000	98 %	2000	98 %	2003	*	*	*
IT0522A	Monza 301528	Traffic	Urban	2000	*	2000	*	2003	*	*	*
IT0593A	Pero 301533	Traffic	Urban	2000	*	2000	*	2003	*	*	*
IT0705A	Verziere 301540	Traffic	Urban	2000	98 %	2000	97 %	2003	98 %	*	*
IT0770A	Arese 301505	Traffic	Urban	2000	*	2000	*	2003	96 %	*	*
IT0777A	Merate 301303	Traffic	Urban	2000	89 %	2000	89 %	2003	12 %	*	*
IT0995A	Cormano1 301513	Traffic	Urban	2000	*	2000	*	2003	*	*	*
IT1016A	Senato Marina 301537	Traffic	Urban	2000	96 %	2000	95 %	2003	*	*	*
IT1035A	Vimercate 301543	Traffic	Urban	2000	*	2000	*	2003	96 %	*	*
PARIS											
FR0332A	Bobigny	Background	Suburban	2000	94 %	*	*	2001	94 %	2002	98 %
FR0346A	Versailles	Background	Suburban	2000	89 %	*	*	2001	*	2002	*
FR0351A	Vitry-sur-Seine	Background	Suburban	2000	96 %	*	*	2001	86 %	2002	77 %
FR0894A	Argenteuil	Background	Suburban	2000	93 %	*	*	2001	*	2002	*
FR0899A	Saint-Denis	Background	Suburban	2000	94 %	*	*	2001	*	2002	*
FR0913A	Garches	Background	Suburban	2000	97 %	*	*	2001	*	2002	*
FR0914A	Ivry-sur-Seine	Background	Suburban	2000	92 %	*	*	2001	*	2002	*
FR0916A	Montgeron	Background	Suburban	2000	99 %	*	*	2001	*	2002	*
FR0923A	Evry	Background	Suburban	2000	98 %	*	*	2001	*	2002	*
FR0327A	Issy-les-Moulineaux	Background	Urban	2000	99 %	*	*	2001	87 %	2002	*
FR0331A	Paris 18θme	Background	Urban	2000	99 %	*	*	2001	93 %	2002	*
FR0337A	Paris 12θme	Background	Urban	2000	93 %	*	*	2001	91 %	2002	*
FR0340A	Neuilly-sur-Seine	Background	Urban	2000	98 %	*	*	2001	*	2002	*

Table B1 Monitoring stations used for comparisons between observations and model result, cont.

	Station name	Station type	Type of area	NO_2		NO_x		PM_{10}		$PM_{2.5}$	
				year	**data.av.**	**year**	**data.av.**	**year**	**data.av.**	**year**	**data.av.**
FR0341A	Aubervilliers	Background	Urban	2000	96 %	*	*	2001	*	2002	*
FR0885A	Gennevilliers	Background	Urban	2000	95 %	*	*	2001	68 %	2002	93 %
FR0886A	Cachan	Background	Urban	2000	78 %	*	*	2001	*	2002	*
FR0892A	Paris 13ème	Background	Urban	2000	96 %	*	*	2001	*	2002	*
FR0900A	Paris 7ème	Background	Urban	2000	94 %	*	*	2001	*	2002	*
FR0918A	Paris 6ème	Background	Urban	2000	90 %	*	*	2001	*	2002	*
FR1181A	Les Ulis	Background	Urban	2000	*	*	*	2001	*	2002	*
FR0335A	Place Victor Basch	Traffic	Urban	2000	88 %	*	*	2001	25 %	2002	*
FR0347A	Avenue des Champs Elysıes	Traffic	Urban	2000	98 %	*	*	2001	*	2002	*
FR0905A	Rue Bonaparte	Traffic	Urban	2000	97 %	*	*	2001	*	2002	*
FR0910A	Quai des Cılestins	Traffic	Urban	2000	84 %	*	*	2001	*	2002	*
PRAGUE											
CZ0009A	Pha8-Kobylisy	Background	Suburban	2000	98 %	2000	98 %	2000	99 %	*	*
CZ0015A	Pha6-Veleslavin	Background	Suburban	2000	95 %	2000	95 %	2000	98 %	*	*
CZ0020A	Pha4-Libus	Background	Suburban	2000	94 %	2000	92 %	2000	91 %	*	*
CZ0010A	Pha2-Riegrovy sady	Background	Urban	2000	96 %	2000	95 %	2000	98 %	*	*
CZ0021A	Pha6-Santinka	Background	Urban	2000	98 %	2000	98 %	2000	100 %	*	*
CZ0008A	Pha1-nam. Republiky	Traffic	Urban	2000	99 %	2000	99 %	2000	100 %	*	*
CZ0011A	Pha5-Mlynarka	Traffic	Urban	2000	98 %	2000	98 %	2000	99 %	*	*
CZ0012A	Pha10-Pocernicka	Traffic	Urban	2000	99 %	2000	99 %	2000	100 %	*	*
CZ0013A	Pha10-Vrsovice	Traffic	Urban	2000	96 %	2000	96 %	2000	98 %	*	*
CZ0014A	Pha4-Branik	Traffic	Urban	2000	97 %	2000	97 %	2000	98 %	*	*
CZ0065A	Pha5-Smichov	Traffic	Urban	2000	98 %	2000	98 %	2000	92 %	*	*
ROME											
IT0953A	Villa Ada 1205820	Background	Urban	2000	92 %	2000	59 %	2000	71 %	*	*
IT0825A	C.so Francia(closed) (3)	Traffic	Urban	2000	93 %	2000	90 %	2000	*	*	*
IT0826A	P.zza e.Fermi 1205813	Traffic	Urban	2000	93 %	2000	93 %	2000	95 %	*	*
IT0827A	L.go Arenula 1205809	Traffic	Urban	2000	40 %	2000	40 %	2000	*	*	*
IT0828A	L.go Magna Grecia 1205810	Traffic	Urban	2000	91 %	2000	91 %	2000	89 %	*	*
IT0887A	Guidonia 1205808	Traffic	Urban	2000	*	2000	*	2000	*	*	*
IT0946A	L.go Montezemolo 1205811	Traffic	Urban	2000	93 %	2000	93 %	2000	*	*	*
IT0954A	V.Tiburtina 1205819	Traffic	Urban	2000	90 %	2000	90 %	2000	*	*	*
IT0956A	Cinecittî 1205804	Traffic	Urban	2000	92 %	2000	91 %	2000	*	*	*
IT1176A	Largo Perestrello 1205875	Traffic	Urban	2000	72 %	2000	71 %	2000	*	*	*
IT1185A	Libia 1205876	Traffic	Urban	2000	95 %	2000	95 %	2000	*	*	*
STUTTGART											
DE0640A	DEBW026:Plochingen	Background	Suburban	2000	99 %	2000	98 %	2002	51 %	*	*
DE0644A	DEBW024:Ludwigsburg	Background	Suburban	2000	100 %	2000	99 %	2002	51 %	*	*
DE0749A	DEBW034:Waiblingen	Background	Suburban	2000	100 %	2000	99 %	2002	50 %	*	*
DE0900A	DEBW042:Bernhausen	Background	Suburban	2000	99 %	2000	98 %	2002	49 %	*	*
DE0621A	DEBW013:Stuttgart Bad Cannstatt	Background	Urban	2000	99 %	2000	98 %	2002	50 %	*	*
DE0637A	DEBW025:Esslingen	Background	Urban	2000	99 %	2000	*	2002	51 %	*	*
DE0748A	DEBW035:Böblingen	Background	Urban	2000	99 %	2000	99 %	2002	50 %	*	*
DE0624A	DEBW011:Stuttgart-Zuffenhausen	Traffic	Urban	2000	100 %	2000	99 %	2002	51 %	*	*
DE1171A	DEBW099:Stuttgart-Mitte-Straie	Traffic	Urban	2000	98 %	2000	97 %	2002	98 %	*	*

Table B1 Monitoring stations used for comparisons between observations and model result, cont.

	Station name	Station type	Type of area	NO_2		NO_x		PM_{10}		$PM_{2.5}$	
				year	data.av.	year	data.av.	year	data.av.	year	data.av.
THESSALONIKI											
GR0045A	Neochorouda	Background	Suburban	2001	72 %	2001	72 %	2001	*	*	*
GR0047A	Panorama	Background	Suburban	2001	99 %	2001	99 %	2001	99 %	*	*
GR0018A	Agia Sofia	Traffic	Urban	2001	96 %	2001	96 %	2001	91 %	*	*
GR0044A	University	Traffic	Urban	2001	97 %	2001	97 %	2001	*	*	*
Not in airbase	Eptapyrgio	Background	Urban	2000	91 %	2000	91 %	2001	88 %	*	*
Not in Airbase	Venizelou	Traffic	Urban	2001	90 %	2001	90 %	2001	95 %	*	*

Note: * = data not available
The percentage indicates the data availability.

Annex C

Emissions calculations

C1 Urban scale

Urban emissions were calculated according to two emission control scenarios, LGEP-CLE and LGEP-MFR (Cofala *et al.*, 2005).

- The CLE (or Current Legislation) scenario includes all known policies that have been implemented by the end of 2003 (or are in the pipeline).
- The MFR (or Maximum Feasible Reductions) scenario includes only those measures that do not require retirement of existing equipment before the end of its technical life time.

C1.1 Outline of methodology

Through personal communication with J. Cofala, sectoral emissions (in kt) were obtained for the aforementioned scenarios and for the years 2000, 2010, 2020 and 2030. Since information of this type was only available at country level and not at city level, the overall country emissions were considered. Emission reductions were then calculated for each country (AT, BE, CZ, DK, FI, FR, DE, GR, HU, IT, PL, PT, ES, UK), year (2010, 2020 and 2030), SNAP category (SNAP 1 to 10 as described in Table C1) and pollutant (NO_X, VOC, SO_2, NH_3, PM_{10} and $PM_{2.5}$) ([4]).

In order to obtain emission reductions at urban level, each country's emission reductions at city level was considered to be equal to the emission reductions at country level. To derive future urban emissions for the 20 urban areas considered, the aforementioned reduction factors were applied to the gridded city emissions (5*5 km^2) (MERLIN emissions). Thus, the future (2030) emissions for each city were produced for the LGEP-CLE and LGEP-MFR scenarios.

At the moment, city growth assumptions are not known. It is reasonable to assume that city growth is equivalent to overall country growth. However, this may lead to incompatibility with actual growth rates for individual cities.

C2 Local scale

C2.1 Outline of methodology

Vehicle fleets originating from the TRENDS model (Giannouli *et al.*, 2005) for each EU15 country were used in order to calculate NO_X and $PM_{2.5}$ emissions for the reference year (2000). The COPERT model was used (Ntziachristos *et al.*, 2000) for a narrow street canyon, which was assumed to have an average daily traffic of 20 000 vehicles per day. For the three non-EU-15 countries (Hungary, Poland and Czech Republic) vehicle fleets extracted from

Table C1 Description of the SNAP sectors considered

SNAP number	Sector description
SNAP 1	Combustion in energy and transformation industries
SNAP 2	Non-industrial combustion plants
SNAP 3	Combustion in manufacturing industry
SNAP 4	Production processes
SNAP 5	Extraction and distribution of fossil fuels and geothermal energy
SNAP 6	Solvent use and other product use
SNAP 7	Road transport
SNAP 8	Other mobile sources and machinery
SNAP 9	Waste treatment and disposal
SNAP 10	Agriculture

([4]) The emission reductions per city and SNAP sector for the year 2030 are available upon request form the authors.

the TREMOVE model version 2.23 (De Ceuster *et al.*, 2005) were used. However, in the case of the three countries there is a lack of reliable input data for the year 2000. For that reason, new vehicle fleets for the year 2000 were re-calculated by extrapolating into the past vehicle population data of future years (2003 onwards) and the COPERT model runs were repeated. For PM_{10} emissions, the European Phenomenology report was used (Putaud *et al.*, 2003) to calculate an approximate value of the $PM_{2.5}$/ PM_{10} mass concentration ratio. This was carried out in order to convert the aforementioned $PM_{2.5}$ emissions to PM_{10}. This factor was differentiated from city to city wherever possible. For cities which were not included in the aforementioned report, the assumption was made that the factor of the city located closest was valid. Moreover, in the case of two-wheelers, updated NO_X and PM emission factors were used. These were produced by LAT (LAT, 2004). The remaining parameters (vehicle speed, HDV percentage etc.) were obtained using the typology methodology for the urban canyons (van den Hout, D. and Teeuwisse, S., 2004).

Generalised attenuation factors were then calculated for NO_X and PM emissions. These attenuation factors were obtained by the following method: Vehicle activity data (1995–2020) from the TREMOVE model version 2.23 ([5]) were inserted in the TRENDS model. Then emission results were calculated using the COPERT III model. In order to produce emission estimates for the scenarios considered (see Section C2.2), suitable emission reductions based on the introduction of Euro V and Euro VI vehicles (for LDVs and HDVs respectively) were applied to the emissions calculated by COPERT. The emission estimates produced were then extrapolated up to the year 2030 and attenuation factors were calculated for the year 2030.

New street emissions were calculated for the street canyons located in the 20 urban areas considered till 2030. This was carried out by applying the above attenuation factors to the reference year emissions. The temporal distribution of the emissions assumed to be valid for the year 2000 was also assumed to be valid for the year 2030.

C2.2 Emission control scenarios

The focus was on additional traffic related measures, which may be enforced in order to reduce air pollutant emissions from road traffic. Measures are specified with a view particularly to the relevance and possibility of urban interventions.

The emission standards currently under discussion at EU level (European Commission, 2004) are considered here for Euro V and Euro VI for light and heavy duty vehicles respectively.

Several different scenarios were run using data estimated by TRENDS with input traffic activity data originating from TREMOVE baseline. These scenarios focus on NO_X and PM emissions. Table C2 and
Table C3 show the emission standards adopted for each scenario (package), and for NO_X and PM emissions respectively. With regard to PM, the actual reduction used was 90 % for the cases suggested with diesel, particulate filtres (DPF) as the technical measure. This can be justified by the fact that if a DPF is used to satisfy a legal limit its reduction effect in real life might go far beyond the legal limit.

For simplicity, one single effective date for new technologies was assumed: 1/1/2011 for Euro V technology with respect to both passenger cars and light duty trucks and 1/1/2013 for Euro VI heavy-duty vehicles and buses.

For the purpose of this study, only the results of the base case scenario and the 'stricter' of the aforementioned packages (package 5 for NO_X emissions and package 3 for PM emissions) were used. The base case scenario of TREMOVE v2.23 was considered to approximate a 'business as usual' scenario corresponding to the LGEP-CLE scenario (see Annex C1). In addition, package 5 and package 3 for NO_X and PM emissions respectively represent the maximum reductions achievable through emission control measures. These are consistent with the specifications set for NO_X and PM emissions in the LGEP-MFR scenario. Finally, emission results were extrapolated up to the year 2030, according to the two scenarios for the time period 2011–2020.

([5]) The TREMOVE model was considered in this study in order to ensure compatibility with the scenarios of CAFE. However, at the present time the TREMOVE model is not finalised and the results of the final version, which were extracted from the latest available version (v 2.23), may vary from those presented here.

Table C2 Reduction percentage of NO_x emissions with respect to Euro IV (for PC and LDV) and to Euro V (for HDV) for Euro V (for PC and LDV) and Euro VI (for HDV) compliant vehicles, according to the various scenarios

	PC — LDV Gasoline	PC — LDV Diesel	HDV
Package 1	–	– 20 %	– 50 %
Package 2	–	– 20 %	– 85 %
Package 3	–	– 40 %	– 85 %
Package 4	– 40 %	– 20 %	– 85 %
Package 5	– 40 %	– 40 %	– 85 %

Table C3 Reduction percentage of PM emissions with respect to Euro IV (for PC and LDV) and to Euro V (for HDV) for Euro V (for PC and LDV) and Euro VI (for HDV) compliant vehicles, according to the various scenarios

	PC — LDV Gasoline	PC — LDV Diesel	HDV
Package 1	–	– 50 %	– 0 %
Package 2	–	DPF	– 0 %
Package 3	DPF (GDI)	DPF	DPF

Table C4 Local scale emissions per city for the reference year (2000) and the year 2030 (CLE and MFR scenarios) for a street with 20 000 vehicles/day

	Reference		CLE		MFR	
	PM_{10}	NO_x	PM_{10}	NO_x	PM_{10}	NO_x
	(g/km/day)		(g/km/day)		(g/km/day)	
Antwerp	2 993	20 420	1 030	10 440	147	6 537
Athens	1 629	26 066	347	13 691	209	7 753
Barcelona	2 469	27 096	904	13 663	325	8 303
Berlin	2 247	18 883	834	8 304	451	5 995
Brussels	2 993	20 420	1 030	10 440	147	6 537
Budapest	1 912	30 146	–	–	–	–
Copenhagen	1 955	26 110	1 139	15 894	85	5 621
Gdansk	2 256	33 866	–	–	–	–
Graz	3 058	20 856	1 547	10 980	87	5 996
Helsinki	1 827	23 606	702	12 337	195	5 801
Katowice	2 256	33 866	–	–	–	–
Lisbon	3 059	24 856	1 363	17 718	538	7 292
London	1 247	23 542	603	10 861	36	3 659
Marseilles	1 993	25 287	931	13 548	220	7 386
Milan	1 653	21 949	443	9 626	133	5 859
Paris	1 993	25 287	931	13 548	220	7 386
Prague	2 175	24 179	–	–	–	–
Rome	1 653	21 949	443	9 626	133	5 859
Stuttgart	2 247	18 883	834	8 304	451	5 995
Thessaloniki	1 629	26 066	347	13 691	209	7 753

Annex D

Table D1 Average yearly wind speeds considered in each city (6)

City	Wind speed (m/s)
Antwerp	3.1
Athens	3.07
Barcelona	2.29
Berlin	2.83
Brussels	3.06
Budapest	2.27
Copenhagen	3.68
Gdansk	3.44
Graz	2.67
Helsinki	3.15
Katowice	2.62
Lisbon	3.13
London	3.74
Marseilles	2.7
Milan	1.66
Paris	2.88
Prague	2.63
Rome	2.5
Stuttgart	2.48
Thessaloniki	1.9

(6) Wind roses indicating the prevailing wind direction for each city are available from the authors upon request.

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