Urban Air Quality and Transport Report

Assessment of the local contribution to air pollution at Urban Hotspots

ETC/ACC Technical Paper 2008/10
December 2008

Nicolas Moussiopoulos, Zissis Samaras, Fotios Barmpas, Liana Kalognomou, Christos Vlachokostas, Leonidas Ntziachristos, Giorgos Mellios and Marina Kousoulidou

European Topic Centre on Air and Climate Change

The European Topic Centre on Air and Climate Change (ETC/ACC)
is a consortium of European institutes under contract of the European Environmental Agency
PBL UBA-D UBA-V NILU AEAT AUTh CHMI MET.NO ÖKO TNO REC

Front page picture:
A compilation of photos by digitalvision® emphasizing the effects of the local scale contribution to air pollution in urban areas.

DISCLAIMER

Foreword

This report presents the work performed within the task "Assessment of the local contribution to air pollution at urban hotspots" of the European Topic Centre on Air and Climate Change (ETC/ACC) 2007 work programme by the Aristotle University of Thessaloniki. In following up on the findings of the EEA's 2006 Technical Report 1/2006, 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 this purpose, the Clean Air for Europe (CAFE) program's Current Legislation (CLE) and Maximum Technically Feasible Reductions (MFR) scenarios were considered on the basis of the reference year 2000, having as a target the year 2020.

Authors
Nicolas Moussiopoulos, Zissis Samaras, Fotios Barmpas, Liana Kalognomou, Christos Vlachokostas, Leonidas Ntziachristos, Giorgos Mellios and Ms Marina Kousoulidou

Acknowledgements

This report was prepared by the European Environment Agency and its European Topic Centre on Air and Climate Change. The EEA project manager was Peder Jensen and the ETC/ACC task leader was Nicolas Moussiopoulos. The important comments and suggestions by Peder Jensen, Anke Lukewille and Jaroslav Fiala as well as other staff of EEA and DG Environment in the final preparation phase of this report are gratefully acknowledged. Many thanks also to the national focal points and other country representatives for their useful comments.

Contents

Executive summary

One of the major factors which adversely affect air quality in an urban area is traffic-related air pollution, with profound negative effects on human health. Alarming evidence of these impacts is continuously emerging. At the same time the degraded air quality in most densely populated urban areas results to increased exposure of the population to particularly hazardous pollutants such as fine particulate matter ($PM_{2.5}$). In most cities, low air quality is the result of increased traffic emissions combined with inadequate ventilation of the streets. Therefore, specific measures for the improvement of air quality through the reduction of traffic emissions are necessary.

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 future air quality trends, the Clean Air for Europe (CAFE) program's Current Legislation (CLE) and Maximum Technically Feasible Reductions (MFR) scenarios were considered on the basis of the reference year 2000, having as a target the year 2020. Two additional scenarios were performed in a limited number of cities, one examining the effect of MFR retrofits for PM control and one on NO_x control of current vehicles

To a large extent, the current report follows up on the findings of the EEA's 2006 Technical Report 1/2006. The numerical methodology applied for the quantification of the air pollution levels has largely adopted the methodology which was applied during the 2006 study. In all, urban background concentrations were calculated for the 20 European cities using the urban scale model OFIS, which made use of regional background concentration levels from the EMEP model results. The contribution of the traffic emissions to the air pollution levels was assessed on the basis of numerical results with the street scale OSPM model. Street level concentrations were calculated for a hypothetical narrow street canyon configuration assumed to be located in the centre of the city.

As regards the traffic emissions, the vehicle technology classifications considered in the calculations were obtained from the latest version of the TREMOVE model (v2.5 – May 2007). The fleet comprised of up to Euro 2 light duty vehicles and Euro II heavy duty vehicles for the reference year 2000. The CLE scenario considered up to Euro 4 light duty and Euro IV heavy duty vehicles in 2020. The MFR scenario considered up to Euro 6 light duty and Euro VI heavy duty vehicles in 2020.The 'retrofit' scenarios were executed using the MFR fleet classification and considered an extended use of aftertreatment devices to older vehicles (e.g. Euro 3) through retrofits. As emissions are calculated for an urban street canyon, it was assumed that large heavy duty vehicles would not use this street. Hence, the fleet of vehicles with a gross vehicle weight of over 20 t was considered zero.

With regard to the scenario selection, it should be clarified that more advanced technologies can be today considered as MFR technologies, such as hybrid or pure electric vehicles, including fuel cell powered cars. This report projects emissions to 2020 and the penetration of these advanced technologies should still be considered limited within this time frame. Therefore, the MFR approach followed in this report represents a conservative but realistic scenario for the development of the vehicle technology mix to 2020. Moreover, the CLE scenario looks outdated, considering that Euro 5 and 6 technologies for passenger cars and Euro V technology for heavy duty vehicles have already been decided at the time of finalising this report (mid 2008). However it has been retained for consistency to the CAFÉ assumptions.

Exhaust emissions of NO_x and PM were calculated using COPERT 4 (v3.0.0 – November 2006). A number of assumptions had to be made on Euro VI heavy duty vehicle emission factors since no emission standard proposal from the European Commission was available at the time of preparation of this report. The Retrofit scenarios were modeled by first considering the use of 95% efficient diesel particle filters to control PM. Then, selective catalytic reduction devices were considered as retrofits for the reduction of NO_x emissions. These were assumed to lead to 50% NO_x reductions. Also, non-exhaust $PM_{2.5}$ and PM_{10} emissions were calculated from tyre and break wear, using the EMEP/CORINAIR Air Emissions Inventory Guidebook methodology.

The ratio of NO_2 over NO_x in the exhaust of different vehicle technologies was considered in detail. Different combustion and aftertreatment technologies lead to NO_2/NO_x ratios that may range from 2% to over 70%. This means that the primary NO_2 emitted may substantially differ between different vehicle technologies, despite these may fulfill the same NO_x emission standard. The NO_2/NO_x ratio for vehicle technologies up to Euro 4 was based on relevant data used in European studies. New ratios were developed for future emission standards on the basis of the technology expected to be introduced.The improved emissions methodology compared to the EEA Technical Report No 1/2006 hinders the direct comparison of the results between the current and the former study. There are three main deviations between the two studies. First, the technology mix in this study was obtained from the latest TREMOVE version compared to the older TRENDS model in the earlier report. TREMOVE projected a lower diesel penetration in some countries (e.g. Denmark) compared to TRENDS. Second, the inclusion of gasoline exhaust PM and non-exhaust $PM_{2.5}$ and PM_{10} emission information in the current report led to a much more refined and precise PM emission inventory. Third, the inclusion of detailed NO_2/NO_x ratios demonstrated that later technology vehicles may emit higher primary NO_2 emissions compared to older ones, which again led to higher NO_2 emissions than earlier expected.

For all cities that were considered, air quality projections for the CLE and the MFR scenarios for the target year 2020 were compared against air quality estimates for the reference year 2000. In all cases, the contribution of traffic to the air pollution decreased significantly with the maximum reductions in the MFR scenario. Air quality projections made for the retrofit scenarios show significant reduction only for NO_x and only marginal for PM. Concentration exceedances of the 2010 limit values were also analysed for all scenarios. Results show a significant reduction in the total number of exceedances especially for NO_2 for all scenarios for the year 2020. However, for PM_{10} for the CLE scenario, the maximum allowed number of 7 daily exceedances per year was violated in 12 cities. On the other hand, for the MFR scenario, most cities show close to zero exceedances. The effect of the aftertreatment technology assumed by the Retrofit scenario on the estimated number of exceedances was negligible.

An analysis of the estimated impacts of the air quality on human health was performed using chronic mortality and chronic bronchitis as the main health endpoints. The analysis was based on the numerical predictions from the air pollution levels for the target year 2020 for the CLE and MFR scenarios. Assessment of the health impacts was performed using selected Exposure-Response Functions (ERFs) that correlate PM_{10} and $PM_{2.5}$ concentrations with the specific health endpoints. Results showed that compared to the reference year 2000, for both scenarios for the target year 2020, there was a considerable reduction of the estimated health impacts due to the long term exposure to PM_{10} and $PM_{2.5}$. The average reductions of the health impacts for the two scenarios averaged over the 20 cities under consideration ranged:

- from a minimum of 20.5% for CLE to a maximum of 36% for MFR based on the long term exposure of PM_{10}
- from a minimum of 37% for the CLE scenario to a maximum of 67% for the MFR scenario based on long term exposure to $PM_{2.5}$

The analysis showed that due to the long term exposure to PM_{10}, the total loss of Life Expectancy (LE) in terms of Years of Life Lost (YOLL) for the 20 cities under consideration is approximately 300000 YOLL for the reference year 2000, 238000 for 2020 CLE and 192000 for 2020 MFR. The corresponding loss of LE in terms of YOLL due to the long term exposure to $PM_{2.5}$ was approximately 215000 YOLL for 2000, 135000 for 2020 CLE and 72000 for 2020 MFR. Due to the long term exposure to PM_{10}, the total estimated chronic bronchitis cases for the 20 cities was approximately 19800 cases for 2000, 15700 for the CLE and 12700 for the MFR scenario for the year 2020.

Similarly, the total estimated number of chronic bronchitis cases due to long term exposure to $PM_{2.5}$ for the 20 cities under consideration is approximately 14400 cases for 2000, 9000 for the CLE and 4750 for the MFR scenario for the target year 2020.

1 Introduction

The main aim of the present study is to assess both the evolution of air quality and the local contribution to air pollution at urban hotspots across Europe in the year 2020. More specifically, it aims to provide estimates for the expected air pollution levels at local scale hotspots within cities, as compared to the urban background concentration levels on the basis of the two scenarios developed within the Clean Air for Europe (CAFE) program (CAFE, 2007), namely the baseline Current Legislation (CLE) scenario and the Maximum Technically Feasible Reductions (MFR) scenario (Amann *et al.*, 2004). The two scenarios for the year 2020 are consistent with the corresponding RAINS 2020 scenarios (version Nov. 2004) which are available on IIASA (2007). In addition, for a limited number of cities two additional scenarios have been investigated. The objective of these two scenarios was to explore the effect of additional measures that local communities can introduce on air quality. The first scenario assumed maximum reductions for Particulate Matter (PM) by retrofitting currently existing technologies with diesel particle filters, while the second one assumed maximum reduction for nitrogen dioxide (NO_2) by retrofitting with NO_x control aftertreatment.

This report follows up on the findings of a 2006 study to project the evolution of air quality in typical urban hotspots across Europe up to the year 2030 (EEA, 2006.). The 2006 study used the expertise gained during the Street Emissions Ceilings project (SEC) within the work programme of the European Environmental Agency's (EEA) European Topic Centre on Air and Climate Change (ETC/ACC) (ETC/ACC Technical Paper 2003/11 and ETC/ACC Technical Paper 2004/5), in order to provide an estimate of hotspot air pollution levels that occur at local scale within cities, as compared to the urban background concentration levels. This report provided general conclusions related to the trends of air quality in the future and summarized results for the CLE and MFR scenarios for the year 2030. The major pollutants that were taken into account were NO_2, NO_x, fine Particulate Matter ($PM_{2.5}$) and PM_{10}. The results clearly indicated that according to both scenarios considered, considerable reductions in the air pollution levels for the year 2030 are expected. However, the major findings of the 2006 study for the air quality projections at street level across Europe can be summed up as follows:

- At street level, the annual 2010 limit value[1] for NO_2 will be met in only very few cities for the CLE scenario and in most cities for the MFR scenario

- The indicative limit value for PM_{10} is not expected to be met in any city, even in the MFR scenario

- For NO_2, the permitted number of exceedances according to the 2010 hourly limit value is expected to be met in all cities for both scenarios. However, exceedances of the PM_{10} indicative limit 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

Moreover, during that study it became obvious that the projection of emissions and hence of the predicted concentration levels at urban hotspots for the year 2030 is bound to large uncertainties. The main reason is that the road transport sector is a highly dynamic source of emissions with a fleet size,

[1] According to Directive 1999/30/EC, in 2010 the limit values to be met for NO_2 are 40 μg/m^3 (annual average) and 200 μg/m^3 (hourly average not to be exceeded more than 18 times a year) whereas for PM_{10} the indicative limit values are 20 μg/m^3 (annual average) and 50 μg/m^3 (daily average not to be exceeded more that 7 days a year). At the time of the completion of the report, the New Air Quality Directive (2008/50/EC) was not yet finalised and comparison against new limit values related to $PM_{2.5}$ and the consideration of the time extensions provided for meeting the NO_2 and PM_{10} limit values granted by the EC under conditions, have not been considered.

fuel mix use and technology and emissions factors rapidly changing with each new regulation step, typically appearing every 4 to 5 years.

Compared to the 2006 study, the current work has now been expanded to include cities in non-EU15 countries, as more reliable data for these countries has now become available. A total of twenty cities have been considered, including four from non-EU15 countries. The modeling methodology that was applied for the assessment of the air quality at urban hotspots largely follows the methodology used for the 2006 study. Projections for the air pollution levels at 20 European cities in 2020 were made, using the complete regional-urban-local scale model application, which consists of:

- The urban scale – OFIS model (Arvanitis and Moussiopoulos, 2003), which is driven by results of the EMEP model (EMEP, 2007) – concentrations and meteorological data – in order to obtain the urban background

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

Additionally, on the basis of the projections for air quality in 2020, it was attempted to asses the impact of air quality on human health in the year 2020 compared to the reference year 2000.

Compared to the 2006 report (EEA, 2006), the current report attempts to better describe the emission evolution and the projection of the local contribution to air quality at urban hotspots across Europe, having the year 2020 as the target. In particular, within the frame of the present study a more realistic emissions estimation of NO_x and PM was made, since between 2006 and 2007 new regulatory steps were decided (Euro 5 and 6 passenger cars and light duty vehicles). These new steps have now been taken into account. Furthermore, the emission projections were also refined with the use of the latest TREMOVE and COPERT versions, to better reflect the effect of new technologies and to better describe the evolution of the fleet mix. Compared to the 2006 report as regards the emissions estimations, the main new elements which were introduced are listed below:

- European vehicle fleets were derived from TREMOVE version 2.5 compared to the 2006 report where a calibrated TRENDS model was used instead

- NO_x emissions factors were extracted from the COPERT 4 software which includes revised estimates of emission performance for late technology vehicles (Euro 4 and later)

- Technology-specific primary NO_2/NO_x ratios were included, in order to account for the significant change of the ratio

- $PM_{2.5}$ emission factor were included for gasoline vehicle exhaust

- Non – exhaust (tyre, break wear) PM were also included which were distinguished between fine and coarse particles

These new elements allow a better characterisation of the effect of relevant policies and policy scenarios on emissions. They also allow addressing questions related to primary NO_2 emission reduction effectiveness and the increasing importance of non-exhaust vs. exhaust PM emissions. Because of the improvements, a direct comparison of absolute emissions levels between the 2006 and the current study is not possible. Furthermore, both the OFIS and the OSPM model predictions for the urban background and the urban hotspot concentrations respectively, are highly sensitive to the local emissions estimation. Therefore, comparison for the local contribution to air pollution across Europe between the years 2020 and 2030 was deemed not possible. Even more, assessment of the impacts of air pollution on human health is highly dependent on the predicted concentration levels. As a result, a comparison between the estimated effects of air pollution levels on health between 2020 and 2030 was also not included.

2 Methodology

This section deals with the methodology adopted for the assessment of air pollution levels at urban hotspots and the corresponding impact on health across Europe. As regards the calculation of the local scale emissions and the numerical predictions for the assessment of air quality at urban hotspots, the methodology largely follows the one applied within the frame of the work performed for the EEA Technical Report No 1/2006 "Air pollution at street level in European cities". In order to benefit from the experience gained in previous works, the methodology is closely related to the work performed within the "Street Emission Ceilings" task of the ETC/ACC during 2003, 2004 and 2006 (task later re-named to "Assessment of the local contribution to air pollution at urban hotspots").

2.1 Emissions methodology

The methodology for the estimation of the urban and local scale emissions largely follows the methodology described in Annex C of the EEA Technical Report No 1/2006 "Air pollution at street level at European cities". In line with the continuous improvements made to the individual elements of the emission estimation methodology, the local emission projections (conducted for the year 2020) have been performed with the latest versions of the TREMOVE and COPERT models, as well as additional new elements to describe the effect of technology on emissions. As analysed in the following sections, the revised and more detailed model characteristics included in these updated versions (e.g. the technology effect on the NO_x/NO_2 split, the revised estimate on NO_x and PM emissions from future technologies, etc.) lead to differences that hinder the direct comparison of local air quality estimates in the years 2020 (presented in this report) and 2030 (presented in EEA Technical Report No 1/2006). A thorough discussion of the emission estimation between the two methodologies is included in this chapter.

2.1.1 Urban scale emissions

Gridded (5×5 km²) 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 the MERLIN project (FP5), using the European Emission model (Friedrich and Reis, 2004; Schwarz, 2002; Wickert, 2001). These inventories were used to estimate background pollutant concentrations and were made available for all 20 urban areas studied per SNAP[2] category, as shown in Table 2.1.

Table 2.1 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

[2] Selected Nomenclature for Air Pollution.

The emission estimates for the year 2020 follow the assumptions of the two Clean Air for Europe (CAFE) scenarios namely the Current Legislation (CLE) and Maximum Technically Feasible Reductions (MFR). The CLE scenario corresponds to the future road transport emission evolution, assuming that no further regulatory steps will be brought, further to the emission standard regulations already in place today. The MFR scenario corresponds to what is expected to be achieved with all legislation that is currently in place or has already been decided for future implementation, and with the foreseeable regulatory steps for the near future. The urban gridded emissions were reduced according to country-specific attenuation factors calculated for each of the SNAP categories shown in Table 2.1.

The country scale emission projections were calculated in the framework of the CAFE project (CAFE, 2007). It is therefore inherently assumed that urban emissions will change in the future in proportion to country emissions. However, urban areas exhibit large changes in their activity level due to infrastructure changes (e.g. new roads, airports, public transport systems, etc.) and population changes (e.g. rapid city growth due to geopolitical reasons or GDP increase). In some countries with developing industrial infrastructure (e.g. new member states), cities have the potential to develop more than the national population growth would call for. On the contrary, countries with diversity in their economic growth sources (such as Scandinavian countries) exhibit a more balanced population growth between urban areas and the countryside. However, in order to adopt a common approach and produce comparable results across cities in Europe and as specific city growth assumptions were not available, the approximation that city growth will be proportional to overall country growth is considered a reasonable approach. In any case, it should be noted at this point that the background concentration levels significantly affect the overall air quality in cities and hence the estimated health impact. However, hot-spot concentrations will not be significantly affected by the exact urban growth and therefore for the needs of the present report this parameter is not considered as deterministic.

2.1.2 Local scale emissions

2.1.2.1 Definition of reference year and scenario fleets

Year 2000 is taken as the reference year for emission calculations. The vehicle classification in emission standards for year 2000 has been taken from the latest version of TREMOVE software (v2.5 – May 2007) for all countries. The year 2000 fleet includes up to Euro 2 passenger car and light duty trucks and up to Euro II heavy duty trucks and buses.

The CLE scenario for the year 2020 is consistent with the RAINS 2020 CLE scenario (version Nov. 2004) which is available on IIASA (IIASA, 2007; CAFE, 2007). It includes up to Euro 4 passenger cars and light duty trucks and up to Euro IV heavy duty vehicles (HDVs). The fleet classification in technologies was again conducted with the latest (v2.5 – May 2007) TREMOVE version. However, all vehicles falling into post-Euro 4 light duty vehicle (LDVs) and post – Euro IV HDVs technologies existing in TREMOVE were all forced to be introduced as Euro 4 and IV respectively in the CLE scenario, in order to be consistent with the technology definitions of this scenario.

The MFR scenario for the year 2020 is consistent with the RAINS 2020 MFR scenario (version Nov. 2004), available on IIASA (2007). It includes up to Euro 6 passenger cars and light duty trucks and up to Euro VI HDVs. The technology classification was taken from TREMOVE v2.5, using the implementation dates for the different technologies suggested in the model. The implementation dates in Tremove are considered as the first of January of the first full calendar year that each technology is in place.

With regard to the scenario selection, it should be clarified that, presumably, more advanced technologies can be today considered as MFR technologies, such as hybrid or pure electric vehicles, including fuel cell powered cars. This report projects emissions to 2020 and the penetration of these advanced technologies is still considered limited. Therefore, the MFR approach followed in this report should be considered as a conservative but realistic scenario for the development of the vehicle technology mix to 2020. Moreover, the CLE scenario looks outdated, considering that Euro 5 and 6 technologies for passenger cars and Euro V technology for heavy duty vehicles have already been decided at the time of finalising this report (mid 2008) but it has been retained for consistency to the CAFE findings.

The ‘retrofit’ scenarios for year 2020 were again executed assuming the 2020 MFR fleets from TREMOVE. Hence, the fleet configuration is exactly the same as in the MFR scenario. The ‘Retrofit’ scenario was assumed to correspond to an option where local (municipal or urban) authorities seek to do more than what emission standards require, in order to solve specific pollution problems (e.g. urban hotspots). In principle, it was assumed that a range of technologies will be retrofitted with advanced aftertreatment systems to examine what additional reductions can be achieved. Two, different options were examined – one referring to NO_x (Selective Catalytic Reduction - SCR retrofits) and one referring to PM emissions (Diesel Particle Filters - DPF retrofits).

2.1.2.2 Fleet and activity data

The share of each vehicle category was derived from the composition of the vehicle fleet in the years 2000 and 2020 (extracted from the TREMOVE software as explained above). On the basis of these shares a new vehicle distribution was calculated for a hypothetical urban street canyon with a traffic intensity of 20 000 vehicles per day, assuming a 7 % share of HDVs. For the street canyon it was assumed that only HDVs (including buses and coaches) with a Gross Vehicle Weight (GVW) lower than 20 tons were allowed, i.e. the share of HDV with a GVW of over 20 tons was set equal to zero.

An average hourly pattern was derived from traffic measurements in various European cities (Athens, Madrid, Cologne, London and Milan) for all vehicle categories, e.g. passenger cars (PC), power two-wheelers etc., as shown in Table 2.2. Based on this pattern the 20 000 vehicles per day were then distributed over the day on an hourly basis.

Table 2.2 Hourly traffic distribution for the various vehicle categories

Category	0:00	1:00	2:00	3:00	4:00	5:00	6:00	7:00	8:00	9:00	10:00	11:00
Gasoline	0.022	0.012	0.007	0.004	0.007	0.022	0.046	0.056	0.052	0.052	0.053	0.055
Diesel PCs	0.020	0.011	0.006	0.004	0.006	0.019	0.048	0.062	0.053	0.052	0.054	0.055
LDV	0.021	0.011	0.006	0.004	0.006	0.020	0.048	0.059	0.053	0.052	0.053	0.055
HDV	0.021	0.011	0.006	0.004	0.007	0.020	0.048	0.058	0.052	0.052	0.053	0.055
Buses	0.019	0.010	0.006	0.004	0.006	0.020	0.049	0.059	0.053	0.052	0.053	0.055
Two wheelers	0.022	0.012	0.007	0.004	0.007	0.022	0.046	0.056	0.052	0.052	0.053	0.055
Category	**12:00**	**13:00**	**14:00**	**15:00**	**16:00**	**17:00**	**18:00**	**19:00**	**20:00**	**21:00**	**22:00**	**23:00**
Gasoline	0.046	0.052	0.056	0.058	0.065	0.065	0.063	0.051	0.044	0.042	0.039	0.031
Diesel PCs	0.045	0.052	0.057	0.060	0.069	0.071	0.066	0.050	0.041	0.038	0.035	0.026
LDV	0.046	0.053	0.057	0.059	0.067	0.069	0.066	0.050	0.042	0.039	0.036	0.028
HDV	0.046	0.053	0.057	0.059	0.067	0.069	0.066	0.050	0.042	0.039	0.036	0.029
Buses	0.046	0.053	0.057	0.059	0.068	0.070	0.067	0.050	0.041	0.039	0.035	0.029
Two	0.046	0.052	0.056	0.058	0.065	0.065	0.063	0.051	0.044	0.042	0.039	0.031

Note: Diesel passenger cars appear more active during the rush hours due to the large contribution of such vehicles to captive fleets (taxis, delivery vehicles, etc.)

2.1.2.3 Emission factors

Exhaust emissions calculations for the reference year (2000) and the 2020 CLE scenario were performed with the latest COPERT 4 model (v3.0.0 – November 2006). No cold start emissions have been included because of the lack of data to accurately estimate the number of starts in the hypothetical street examined. It is reasonable to assume that the number of starts in the area studied is much less than the passing traffic, which further demotes the contribution of cold-starts. In any case, the cold start effect is considered minimal for PM and only marginal for NO_x, since cold start has no strong effect on these two pollutants (cold start is important mainly for HC and CO).

In order to estimate emissions for the MFR scenario, appropriate assumptions had to be made for emission technologies, where no information exists in COPERT. This includes Euro 5 and Euro 6 passenger cars and light duty trucks and Euro VI HDVs. Also, in order to directly assess the effect of regulation on emission levels from HDVs, the Euro V HDV emission factors were also estimated on the basis of a reduction factor over Euro IV (see Table 2.3), despite the fact that emission factors for Euro V HDVs are included in COPERT 4 as well. Table 2.3 presents the implementation years and the emission reductions over Euro 4 (LDVs) and Euro IV (HDVs) of future emission standards considered in this study.

The following clarifications need to be given for the values in Table 2.3. First, with respect to the estimated PM reductions, the applied reduction factor is higher than the reduction considered by the legislation. This is because it is expected that the PM emission reduction required by the legislation will only be reached with the introduction of diesel particle filters (DPFs), which have a proven filtration efficiency of at least 95% in particulate mass. Hence, it is expected that the actual emission reduction to be reached will be higher than that implied by the emission standards.

Table 2.3 Reduction factors over Euro 4 (LDVs) and Euro IV (HDVs) to estimate emissions of future emission technologies in the MFR scenario

Category	Emission Standard	Impl. Year	NO_x Emission Standard (g/km or g/kWh)	NO_x Reduction according to Legislation (%)	NO_x Applied Reductions (%)	PM Emission Standard (g/km or g/kWh)	PM Reduction according to Legislation (%)	PM Applied Reductions (%)
Gasoline PCs	Euro 4	2005	0.08	0	0	-	0	0
	Euro 5	2010	0.06	25	25	-	0	0
	Euro 6	2015	0.06	25	25	-	0	0
Diesel PCs	Euro 4	2005	0.25	0	0	0.025	0	0
	Euro 5	2010	0.18	28	28	0.0045	88	95
	Euro 6	2015	0.08	68	68	0.0045	88	95
Gasoline LDTs (Class II)	Euro 4	2006	0.1	0	0	-	0	0
	Euro 5	2011	0.075	25	25	-	0	0
	Euro 6	2016	0.075	25	25	-	0	0
Diesel LDTs (Class II)	Euro 4	2006	0.33	0	0	0.04	0	0
	Euro 5	2011	0.235	28	28	0.0045	92.5	95
	Euro 6	2016	0.105	68	68	0.0045	92.5	95
HDVs (ETC)	Euro IV	2006	3.5	0	0	0.03	0	0
	Euro V	2009	2	43	43	0.03	0	0
	Euro VI (1)	2014	0.4	89	89	0.010	66	95

(1) Potential standard to be proposed by the European Commission (COM(2007) 851).

The second remark concerns the Euro VI emission standard for HDVs. The proposal from the European Commission (COM(2007) 851) asks for 400 mg/kWh NOx and 10 mg/kWh PM. In principle, these emission values suggest that the application of both DPF and $deNO_x$ (SCR) aftertreatment systems will be required to achieve the Euro VI emission levels and this will lead to substantial reductions of both NO_x and PM.

In the case of the ‘Retrofit’ scenario, additional reductions can be brought on existing vehicle technologies over the MFR scenario, due to the retrofit of aftertreatment systems. Since retrofits are not straightforward for vehicle technologies that lack engine control units, these have been limited to post-Euro 3 passenger cars and light duty trucks and post Euro III HDVs. Two aftertreatment options have been considered. For PM, it is expected that Euro 3 and Euro 4 diesel passenger cars and light duty vehicles can be retrofitted with DPFs that lead to a 95 % reduction over the corresponding emission standard. Also, it is assumed that Euro IV and Euro V HDVs can also be fitted with 95 % efficient DPFs. For NO_x, it is considered that Euro 5 and Euro 6 can be equipped with SCR systems which may lead to 50 % lower emissions than gasoline Euro 4. In this case, a Euro 5-6 gasoline car would roughly correspond to US Tier 2B in 3-4 emission standards. The SCR retrofits are also considered for diesel passenger cars and HDVs. The proposed reductions have been based on experimental information on the effect of after treatment technologies, however, they rather represent a best-case scenario as real-world operation and failures may lead to lower overall efficiencies in practice. However, in order to maximise the potential effect of the retrofit scenario, these values have been retained. In the case of SCR, a gradual reduction is expected as emission standards improve, due to the lower engine-out NO_x emissions (cooled EGR use) of later technologies. The SCR retrofits also lead to very low NO_2/NO_x ratios. Those have been taken as 2% for gasoline and 5% for diesel vehicles. Table 2.4 provides a summary of the technologies and the reductions brought on emissions for each of the two Retrofit scenarios considered.

Table 2.4 Reduction factors considered for the retrofit scenarios

Category	Emission Standard	NOx reductions		PM Reductions	
		Technology	Reduction over Euro 4/IV (%)	Technology	Reduction over corresponding emission standard (%)
Gasoline PCs	Euro 4	-	0	-	0
	Euro 5	SCR	50	-	0
	Euro 6	SCR	50	-	0
Diesel PCs	Euro 3	-	0	DPF	95
	Euro 4	-	0	DPF	95
	Euro 5	SCR	76	As required by the emission standard	0
	Euro 6	SCR	84		0
Gasoline LDTs	Euro 4	-	0	-	0
	Euro 5	SCR	50	-	0
	Euro 6	SCR	50	-	0
Diesel LDTs	Euro 3	-	0	DPF	95
	Euro 4	-	0	DPF	95
	Euro 5	SCR	76	As required by the emission standard	0
	Euro 6	SCR	84		0
HDVs (ETC)	Euro IV	SCR	80	CRT(1)	95
	Euro V	SCR	90	CRT(1)	95
	Euro VI	SCR	89	As required by the emission standard	0

(1) Continuously regenerating trap

The range of combustion and aftertreatment technologies to be applied to reach future emission standards also has significant implications on the NO_2/NO_x ratio in vehicle exhausts. The ratio of NO_2/NO_x in primary emissions is also of importance when estimating urban NO_2 concentrations. In order to address this issue, relevant information was collected from ongoing studies in Europe and from a literature survey on primary emissions. Results are presented in Table 2.5. The two European studies considered are the AEAT (2007) study on behalf of DG ENV and a study run by TNO, concerning primary NO_2 emissions for local scale air quality assessment in the Netherlands (Smit, 2007). Detailed measurement and feasibility studies were used as a source to estimate emissions in advanced aftertreatment systems, such as SCR deNO_x aftertreatment (Mayer *et al.,* 2007; Wiartala *et al.,* 2007).

All PM emissions from vehicle exhaust are considered to fall within the $PM_{2.5}$ size definition. However, road traffic contributes to PM emissions from non-exhaust sources as well. Primary non-exhaust PM includes particles from road, tyre and break attrition, while re-suspension is not considered as a primary but as a secondary source of PM and it is not included in the calculations. This does not mean that re-suspension is a trivial contributor to ambient PM concentrations, in particular in hot-spots. However, this depends on a number of factors, including slit load level on the streets and local morphology that cannot be covered in a wide study as the one presented in this report. Detailed studies with localized interest should take re-suspension into consideration and estimate its contribution. In order to estimate emissions from primary sources, the methodology included in the relevant EMEP/CORINAIR Emission Inventory Guidebook chapter (Ntziachristos and Boulter, 2003) has been applied. This methodology allows to separately estimating $PM_{2.5}$ and PM_{10} fractions for non exhaust PM. As a result, the $PM_{2.5}$ emissions that have been calculated in this study correspond to the sum of exhaust and non exhaust $PM_{2.5}$. The corresponding PM_{10} emissions correspond to the sum of exhaust $PM_{2.5}$ and non-exhaust PM_{10}.

Table 2.5 Primary NO_2/NO_x mass ratios considered in European studies and applied in the different scenarios of this study

Category	Technology	NO_2/NO_x primary mass ratio (%)				
		AEAT Study	TNO Study	Ratios applied in this study		
				CLE	MFR	Retrofit
Gasoline PCs	pre-Euro	4	5	4	4	4
	Euro 1 - Euro 2	4	5	4	4	4
	Euro 3 - Euro 4	3	5	3	3	3
	Euro 5	3	5	-	3	2
	Euro 6	3	-	-	2	2
Diesel PCs	pre-Euro	11	20	11	11	11
	Euro 1 - Euro 2	11	20	11	11	11
	Euro 3	30	40	25	25	25
	Euro 4	55	40-70	55	55	55
	Euro 5	55	70	-	55	5
	Euro 6	55		-	55	5
LPG PCs	pre-Euro	5	5	5	5	4
	Euro 1 - Euro 3		5	5	5	4
	Euro 4		5	5	5	3
	Euro 5		-	-	5	2
	Euro 6		-	-	5	2
Gasoline LDTs	pre-Euro	-	5	4	4	4
	Euro 1 - Euro 2	-	5	4	4	4
	Euro 3 - Euro 4	-	5	3	3	3
	Euro 5	-	5	-	3	2
	Euro 6	-		-	2	2
Diesel LDTs	pre-Euro	-	20	11	11	11
	Euro 1 - Euro 2	11	20	11	11	11
	Euro 3	30	40	25	25	25
	Euro 4	55	40-70	55	55	55
	Euro 5	55	70	-	55	5
	Euro 6	55	-	-	55	5
HDVs (ETC)	pre-Euro	11	10	11	11	11
	Euro I - Euro II	11	10	11	11	11
	Euro III	14	10	14	14	14
	Euro IV	10	10	14	14	5
	Euro V	10	10	-	18	5
	Euro VI	10	-	-	35	5

2.1.2.4 Summary of changes compared to older estimates

Several new calculation elements have been included in this report compared to the Technical Report No 1/2006 in an effort to improve the projection of emissions in the future. In particular, the latest versions of available models (both TREMOVE and COPERT) have been used, as well as the Air Emission Inventory Guidebook AEIG chapter on non-exhaust PM emissions from road-transport. Some new ad-hoc elements have been also introduced to detail the emission information. Table 2.6 summarizes the main differences compared to the estimate conduced in 2006.

Table 2.6 Summary of main differences between the current and previous emission calculation methodologies

Effect	Technical Report 1/2006	Current Work
Fleet Composition	TRENDS model with corrections based on TREMOVE 2.23 (2003 Data)	TREMOVE v 2.5 used (2007 data)
Gasoline exhaust PM Emissions	Zero emission factor	Technology-specific emission factors for gasoline vehicles
Emission Factors	COPERT 3 emission calculations (post Euro 1 emission factors based on extrapolations)	COPERT 4 emission calculations (Measured emission factors up to Euro 4, extrapolations only for Euro 5, 6)
$PM_{2.5}/PM_{10}$ Ratio	Constant (phenomenological) ratio used to derive PM_{10} on the basis of exhaust $PM_{2.5}$ emissions	Different PM2.5 and PM10 emission factors used, including the contribution of non exhaust emissions. The $PM_{2.5}/PM_{10}$ ratio changes in the future.
NO_2/NO_x Ratio	Rough NO_2/NOx ratios were used	Detailed NO_2/NO_x ratios were used as a function of technology, based on available measurements where possible. In particular the use of SCR in trucks reduces the NO_2/NO_x ratio
Euro 5 NO_x	MFR includes Euro 5 diesel PCs and LDVs with 20% NO_x reduction over Euro 4	MFR includes Euro 5 diesel PCs and LDVs with 28% NO_x reduction over Euro 4
Euro 5 PM	MFR includes Euro 5 diesel PCs and LDVs with 90 % PM reduction over Euro 4	MFR includes Euro 5 diesel PCs and LDVs with 95 % PM reduction over Euro 4
Euro 6	MFR does not include Euro 6 for passenger cars	MFR includes Euro 6 for passenger cars (68 % NO_x and 95% PM reduction in diesels over Euro 4)
Euro VI NO_x	MFR includes Euro VI trucks with 85% NO_x reduction	MFR includes Euro VI trucks with 89% NO_x reduction

Due to these changes, the following main deviations between the 2006 study (Technical Report 1/2006) and 2007 study (current report) are expected.

- The use of TREMOVE 2.5 brings updated information regarding the fleet mix (2007 Data) and in particular the share of diesel passenger cars in urban fleets. In some countries (e.g. Denmark) the diesel car share remains lower than what was previously projected and this would lead to a more moderate increase of emissions than earlier projected.

- The use of separate emission factors for $PM_{2.5}$ and PM_{10}, including the contribution of non-exhaust particles and exhaust PM from gasoline vehicles will lead to higher PM emissions in the future than earlier projected.

- The inclusion of detailed NO_2/NO_x ratios which show an increase in primary NO_2 emissions from late vehicle technologies.

The finding that NO_x and in particular NO_2, emissions from diesel passenger cars have not been reduced as effectively as previously expected will also lead to increased NO_x emission projections in the future.

2.2 Local scale contribution to air pollution methodology

The air quality modelling methodology adopted for the estimation of air pollution levels at urban hotspots across Europe for the projection year 2020 is in effect the same methodology which was applied for the estimation of the local contribution to air pollution at hotspot areas across Europe in

2030, presented in the EEA Technical Report No 1/2006. As the methodology is thoroughly described in the aforementioned report, it is not deemed necessary to describe it here in detail and the reader is referred to this report for a comprehensive description. Furthermore, the methodology has already been validated on the basis of NO_2, NO_X, PM_{10} and $PM_{2.5}$ comparison between modelling results and concentration measurements available in Airbase (AIRBASE, 2007) for the reference year 2000. As a result, no further validation of the approach used was performed for the needs of the present report.

The air quality modelling methodology is a multi-scale modelling approach which consists of a complete regional - urban - local - scale model cascade application using three models: EMEP, OFIS and OSPM. The regional scale model EMEP (EMEP, 2007) was applied in the framework of the CAFE programme and the results for the CLE and MFR scenarios (meteorology and concentrations) were used as boundary conditions for the OFIS model (Arvanitis and Moussiopoulos, 2003). This approach was assumed to adequately describe the urban air quality within and close to each city. The street-scale model OSPM (Berkowicz *et al.*, 1997) was applied using urban background concentrations derived from the OFIS model to estimate air quality within particular types of street configurations.

The urban emission inventories required by the OFIS model were calculated through a top-down approach adopted within the MERLIN project (MERLIN, 2007), for the 20 cities under investigation. In compliance with the study performed for the year 2030, the local air quality analysis was performed based on a generic approach for the definition of the test cases (streets), following the Typology Methodology (Moussiopoulos *et al.*, 2004; Moussiopoulos *et al.*, 2005).

Numerical simulations at a street scale were performed using the OSPM model. Local air quality estimates for the year 2020 were performed for a specific street type, located in the city centre in each of the 20 European cities. Due to the absence of a detailed database for street types across Europe (from which the statistically most occurring street could be extracted), a generic approach was adopted. Such an approach was studied by Moussiopoulos et al. (2004) and Moussiopoulos et al. (2005), where a first attempt to categorise street types according to various parameters and parameter ranges is described. In this report, the type of street considered has a small aspect ratio (Width of Street / Average Height of Buildings, W/H) of W/H = 0.5, which represents a narrow canyon (worst case scenario). During the 2006 study (EEA, 2006), a detailed analysis of the dependency of the air pollution levels at urban hotspots on the type of street was performed, in order to investigate the effect of street geometry on dispersion and such an analysis was not included in the current report.

Air quality at urban hotspots was assessed on the basis of calculations for the annual average concentrations and street increments for NO_2, NO_X, PM_{10} and $PM_{2..5}$. Moreover, hourly NO_2 and daily PM_{10} exceedances of the corresponding limits defined within the relevant legislation were also calculated. More specifically, for CLE and MFR, the local contribution to air pollution at street level was assessed based on comparison of numerical results for the annual mean street increments for NO_2, NO_x, PM_{10} and $PM_{2.5,}$ between the reference year 2000 and the year 2020. In addition, comparison of the estimated exceedances of the hourly NO_2 and daily PM_{10} of the 2010 limit values between the two scenarios was also made. An additional comparison of results for the annual mean street increments for all pollutants considered as well as for the hourly NO_2 and PM_{10} 2010 limit values between the MFR and the Retrofit scenarios was also performed.

In order to gain a perspective of the evolution of the overall air pollution levels up to 2020, the individual contribution of the regional (EMEP model), urban (OFIS model) and the local (OSPM model) scales to the annual mean air pollution levels for NO_2 and PM_{10} concentrations across Europe was computed for CLE and MFR scenarios and for a selected number of cities. Such a comparison between the MFR scenario and the Retrofit scenario was not included because both scenarios assume the same regional and urban scales emissions and concentrations.

2.3 Health impacts assessment methodology

Human activities cause damages and impose risks on human beings, ecosystems and materials. A consensus has been emerging among public health experts that air pollution, even at current ambient levels, aggravates morbidity (especially respiratory and cardiovascular diseases) and leads to premature mortality (e.g. Wilson and Spengler, 1996; WHO, 2003; Holland *et al*., 2005a and AIRNET, 2007). The statistical correlations between air pollution and a health impact are called associations. The aforementioned consensus is based on the past decade's epidemiological studies in Europe and worldwide which have measured increases both in mortality and morbidity associated with air pollution (Krzyzanowski *et al.,* 2002). More specifically, state-of-the-art epidemiological research has found consistent and coherent associations between air pollution and various outcomes (e.g. respiratory symptoms, reduced lung function, chronic bronchitis, and premature mortality) (Künzli *et al.,* 2000).

Furthermore, some effects may be related to short-term exposure while others have to be considered contributions of long-term exposure. Although the mechanisms are not fully explained and there is less certainty about specific causes, most recent studies have identified fine particles ($PM_{2.5}$) as a prime culprit; O_3 has also been implicated directly. According to the EternE 2005 report, there may be significant direct health impacts of SO_2. As regards NO_x it has not yet been associated with direct impacts on human health. As evidence of health effects of air pollution has accumulated, European governments, the World Health Organisation (WHO) and other groups have begun to use data from these studies to inform environmental policies through, for example, quantitative estimates of impact of air pollution on public health (Krzyzanowski *et al.,* 2002).

The basic steps in Health Impact Assessment (HIA) comprise of the selection of the set of health outcomes associated with air pollution, the adoption of the risk estimates through Exposure Response Functions (ERFs) on the basis of epidemiological studies and the application of the adopted ERFs for the distribution of the exposure experienced by the target population. The main outcome of the analysis lies in the quantification of the expected health burden owing to the exposure of the target population, expressed in terms of the number of cases or Years of Life Lost (YOLL) attributable to the exposure (Krzyzanowski et al, 2002). This analysis can be extended to include economic valuation of the impact, which is however beyond the scope of the current study. It should be stressed that, whereas all studies before 1996 calculated a number of premature deaths, there has been a growing recognition in recent years that it is more meaningful to look at loss of LE and YOLL (Rabl, 2003, ExternE, 2005). YOLL is a meaningful and appropriate impact indicator for all risk factors, even those that are not observable as the cause of an individual death.

The first step was the selection of the set of health outcomes associated with air pollution. The ERFs adopted in this analysis correlate PM_{10} and $PM_{2.5}$ with chronic mortality and chronic bronchitis. The adopted methodology focused on PM, due to the fact that the most up to date epidemiological studies have shown that the correlation of the PM concentrations with the impact of air pollution on human health is the most consistent correlation between air pollutants and health. In addition, several multi-pollutant analyses have shown that PMs are the most significant pollutants that affect human health. Consequently, the most important damage cost (and the largest contribution to the corresponding total damage cost due to air pollution) is related to chronic mortality due to PM, calculated on the basis of Pope *et al*. (2002) (ExternE, 2005). The term chronic mortality, chosen by analogy with acute and chronic morbidity impacts, indicates that the total or long-term effects of air pollution on mortality are included, in contrast to acute mortality impacts, which are observed within a few days of exposure to air pollution. Another important contribution comes from chronic bronchitis due to particles (ExternE, 2005).

The second step was the adoption of risk estimates. Risk estimates are formulated in terms of Relative Risk[3] (RR) or the respective ERF based on epidemiological studies. An ERF relates the quantity of a pollutant that affects a receptor (or a specific part of the population) to the physical impact on this receptor (e.g. incremental number of hospitalisations). Most epidemiological studies report their

[3] Relative Risk is defined as the ratio of the incidence observed at two different exposure levels.

results in terms of RR. This means that one needs to translate RR in terms of an ERF for the incremental cases per exposure increment in order to quantify damages. Within the frame of the adopted methodology, it is convenient to define all adopted ERFs in terms of YOLL (or cases) per year per "average" person per μg/m^3 since this way they can be applied directly to the entire population. This accommodates the assessment of loss of life expectancy (LE) for chronic mortality and incremental cases of chronic bronchitis for all 20 European cities under consideration with a single ERF per health effect per pollutant.

The unfavorable implications over the health endpoints under consideration that are avoided (or not) after the adoption of either the CLE or the MFR scenarios were estimated through the following expression:

$$\Delta_{cases,i} = R_{i,p} \cdot \Delta conc_P \cdot pop \qquad (1)$$

where:

- $R_{i,p}$: Correlation coefficient between the pollutant's p concentration variation and the probability of experiencing or avoiding a specific health implication i (slope of the ERF)
- $\Delta conc_p$: Change in pollutant's p concentration after the adoption of an emission reduction scenario
- *pop*: population units exposed to pollutant p (target population)

It is therefore evident that the ERFs included in the present analysis were assumed to be linear without a threshold. The linear assumption is valid for health endpoints ERFs (ExternE, 2005). The number of YOLL or cases that are avoided over chronic mortality and chronic bronchitis respectively after the adoption of either the CLE or the MFR scenarios defined the benefit on human health of the adopted scenario. The percentage of YOLL or cases that are avoided is equal to the percentage of the corresponding PM concentration reduction:

$$\frac{\Delta_{cases,i}}{Cases_{reference,i}} = \frac{\Delta conc_P}{Con_{p,reference}} \qquad (2)$$

Equation 2 defines the abatement efficiency (%) of the adopted scenarios in comparison to the reference case (2000) due to concentration variation. This percentage is independent of the slope $R_{i,p}$ and the number of receptors pop of the target population.

It is very difficult and costly to measure the total impacts of air pollution on mortality both in the short and in the long term. However, in recent years, several important epidemiological studies have succeeded in measuring the long-term impacts of air pollution on mortality. Two of these cohort[4] studies (Dockery *et al*., 1993, Pope *et al*., 1995, Pope *et al*., 2002) found positive correlations between exposure to PM and total mortality. Confirmation of long-term mortality impacts was recently provided by a cohort study in the Netherlands (Hoek *et al*., 2002).

The study by Pope *et al* (1995, 2002) used a population sample which consisted of about half a million individuals. Compared to all other long term mortality studies, this sample was the largest ever used. Therefore for the needs of the present analysis, the RR proposed in the aforementioned study was

[4] These studies are called cohort studies because they analyse the survival of a cohort of individuals over a long period, at least several years, and correlate it with individual exposure to air pollution.

adopted. Pope *et al.,* (2002) reported that depending on the assumptions made for the relevant exposure period, for an increase of the concentration of $PM_{2.5}$ by 10 μg/m^3, RR can receive values between 1.04 and 1.06. Assuming an average RR of 1.05 of Pope *et al.* (2002) for every 10 μg/m^3 increase in the concentration of $PM_{2.5}$ and after recalculating LE loss implied by this RR, the methodology of ExternE which was adopted for the needs of the current report, takes the ERF's slope for chronic mortality as:

$$R_{CM,PM_{10}} = 4.0 \cdot 10^{-4} \frac{YOLL}{year \cdot receptor \cdot \mu g / m^3} \tag{3}$$

for PM_{10} applicable to the entire population and

$$R_{CM,PM2.5} = 6.67 \cdot 10^{-4} \frac{YOLL}{year \cdot receptor \cdot \mu g / m^3} \tag{4}$$

for $PM_{2.5}$ applicable to the entire population.

It should be noted at this point that only ages above 30 have been included in the calculations since the underlying cohort studies did not include younger people. However in effect, as regards the adult mortality, age does not pose a limiting factor, since the RR found by Pope *et al*. (2002) and used here is age independent (Krewski *et al*., 2000). The absolute mortality during the period between the end of infancy and the age 30 is very low and hence, any increase due to air pollution would result into a negligible contribution to the total population LE (ExternE, 2005).

Chronic bronchitis is defined as reporting chronic cough or sputum on most days, for a period of at least three months of the year and for at least two years. Using the RR and a background incidence rate of 0.378% from Abbey *et al*. (1995a), Hurley *et al*. (2005a) derived an estimated ERF (ExternE, 2005):

$$R_{CB,PM_{10}} = 2.65 \cdot 10^{-5} \frac{cases}{year \cdot receptor \cdot \mu g / m^3} \tag{5}$$

for PM_{10} applicable to adults (aged 27+) and

$$R_{CB,PM_{2.5}} = 4.42 \cdot 10^{-5} \frac{cases}{year \cdot receptor \cdot \mu g / m^3} \tag{6}$$

for $PM_{2.5}$ applicable to adults (aged 27+). It should be noted, that the population sample considered within the frame of the aforementioned study was aged over 27 years.

Although WHO in its 2005 guidelines uses 0.5, giving PM2.5 a higher relevance, for the needs of the present study, a ratio of 0.6 for $R_{i,PM_{10}} / R_{i,PM_{2.5}}$ was used, in accordance with the ExternE methodology. The adoption of ERFs is followed by the application of this function to the distribution of exposure experienced by the target population. In the present analysis the selected health outcomes (number of cases of chronic bronchitis or YOLL due to chronic mortality) are estimated per million inhabitants per year. This estimation accommodates the present analysis for three reasons:

- It is easy to comprehend and even more convenient to depict the effectiveness of the CLE and MFR scenarios with respect to benefits on human health, when they are normalised to an equal number of exposed receptors for all 20 cities

- The accurate estimation of the number of receptors exposed to air pollution (target population) for all 20 cities both for the reference year 2000 and the target year 2020, formed a prerequisite for the application of ERFs, to the projected distribution of exposure experienced by the target population

- The number of population exposed to air pollution due to traffic contribution at street level is very difficult to estimate inside the core of the city

One of the main difficulties of this work lies in the high uncertainty introduced due to the highly varying conditions at a local level. For most policy applications such detail is not relevant. In the present methodology, as regards the public-health impact of traffic-related air pollution at street level, an annual uniform population density is assumed for all 20 European cities. The calculated annual average street increment at the urban hotspots, allowed an estimation of the increment of health impacts due to traffic-related air pollution at street level on top of the expected health impacts due to the average background concentration at urban level. In addition to the analysis of the contribution of the traffic-related air pollution to the total expected health outcomes, a demonstrative comparison of the health impacts related to the contribution of traffic to air pollution at urban hotspots with respect to every adopted scenario was realised.

3 Emissions

The emissions of NO_x, NO_2, $PM_{2.5}$ and PM_{10} from road transport for the reference year (2000) and the two scenarios studied for year 2020 (CLE and MFR) are presented in this chapter. These data have been used as input to calculate the evolution of the air quality in the cities examined. It is repeated that emissions are calculated for a street canyon section of 1 km with average daily volume of 20000 vehicles. For the production of total emissions, exhaust emissions have been calculated with the latest COPERT 4 version as in detail described in the methodology chapter. However, they do not include cold start emissions, because of the lack of statistical data to accurately estimate the number of starts in the different hypothetical types of streets examined. Cold start overemission is in any case not considered to be important. First because it does not have a big effect on NO_x and PM and second, because the number of starts in a street section of 1 km is comparable negligible to the 20000 passing vehicles considered. Also, PM re-suspension has not been taken into account as this depends on the local air exchange and road slit level conditions, which are not possible to estimate in this study. The graphs presenting $PM_{2.5}$ and PM_{10} emissions include PM emissions from both exhaust and non-exhaust sources (road, tyre and break attrition).

3.1 Total NO_x and NO_2 emissions per category

The evolution of the average total street level emissions of NO_2 and NO_x from the different vehicle categories in all cities up to the year 2020 are illustrated in Figure 3.1. In 2000, gasoline passenger cars and HDVs are the two main sources of NO_x emissions. A reduction appears in the 2020 CLE scenario compared to the reference year and an even larger reduction in the 2020 MFR scenario. The latter is due to the introduction of the Euro 5 and 6 emission standards (both for light duty and HDVs). Although the reduction is very significant for gasoline passenger cars, it does not appear as effective as for diesel cars. There are two reasons for this result. First, the share of diesel cars in the future increases in comparison to gasoline ones, as a result of the measures to reduce CO_2 emissions. Second, the real-world performance of diesel passenger cars appears to lead to much higher NO_x emissions than what the emission standards would call for, even for advanced diesel technologies (COPERT, 2006; Hausberger, 2006), which is one of the findings of the ARTEMIS (2007) project. This indeed seems to be a problem of the effectiveness of the emission standards.

Emissions from power two wheelers seem to marginally increase in the future. Of course, one needs to take into account that power two wheelers are overall negligible contributors to NO_x emissions, hence any marginal increase does not constitute a severe environmental problem. This increase originates from the fact that power two wheelers of conventional technology (the majority of vehicles in 2000) operated on rich fuel mixtures which led to extremely low NO_x emissions (but high HC and CO emissions). The introduction of improved technology (stoichiometric combustion with three way catalyst) may lead to marginal increase of NO_x emissions but a significant reduction in CO and HC. Hence, in the case of power two wheelers the scenarios MFR and CLE mainly refer to HC and CO and not NO_x (or PM).

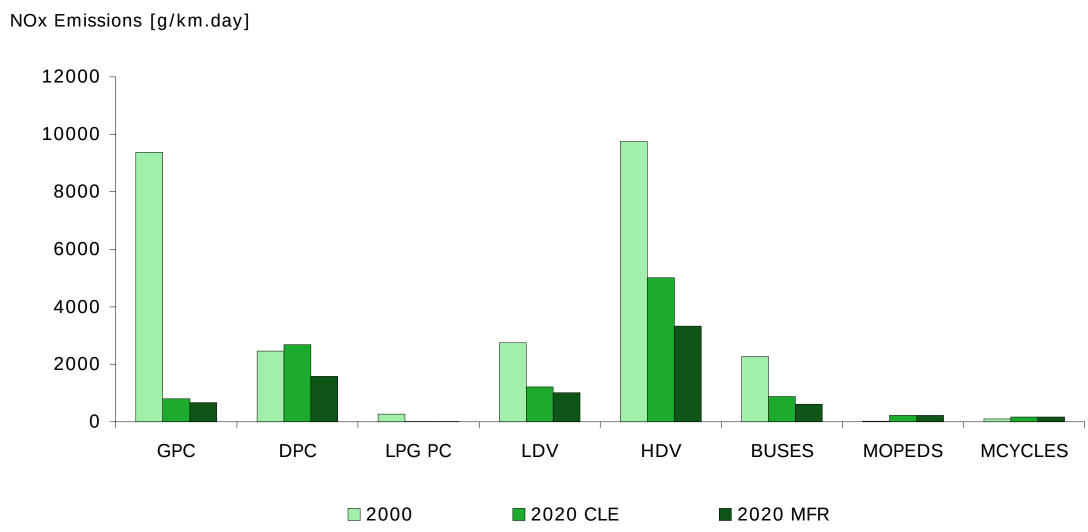
Concerning the average NO_2 street level emissions from all cities, the reference year 2000 presents the highest emissions for all vehicle categories, except again for diesel passenger cars. Similar to the case of total NO_x emissions, this is in part due to the higher share of diesel cars. However, the specific NO_2 emission factors, as explained in the methodology chapter, increase as the diesel aftertreatment technology improves due to the catalytic oxidation of NO to NO_2 in the exhaust. This again may have implications to the attainment of air quality standards in the future.

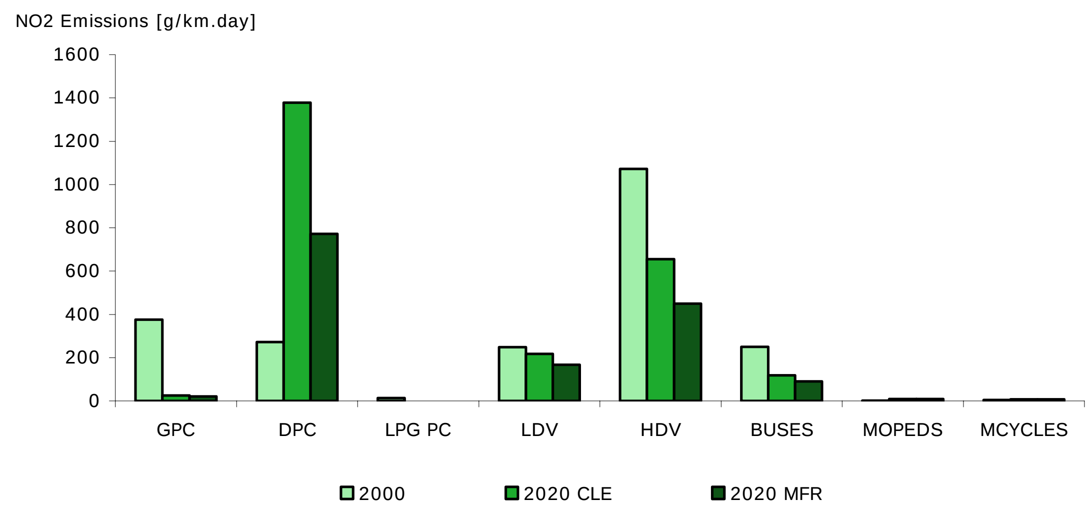
3.2 Evolution of the NO_2/NO_x emissions ratio

The range of combustion and aftertreatment technologies to be applied to reach future emission standards has also significant implications on the NO_2/NO_x ratio in vehicle exhausts. The ratio of NO_2/NO_x in primary emissions is also of importance when estimating urban NO_2 concentrations. In order to address this issue, relevant information was collected from ongoing studies in Europe and from a literature survey on primary NO_2 emissions (Table 2.5). Based on the data collected, Figure 3.2 shows the average emission ratio distinguished per vehicle category and scenario.

In general, the NO_2/NO_x ratio is already low and further decreases for gasoline passenger cars with the upcoming Euro 5-6 technologies. In particular this is due to assumed reduction of NO_2/NO_x for Euro 3 and 4 passenger cars and the assumed use of SCR in Euro 6 passenger cars which leads to further reduction of this ratio.

Figure 3.1 Average (per city) emissions of total NO_x and NO_2 from different vehicle categories per km street segment with 20.000 vehicles/day. Projections are based on expected emission factor reductions for new technologies and changes in the vehicle stock mix

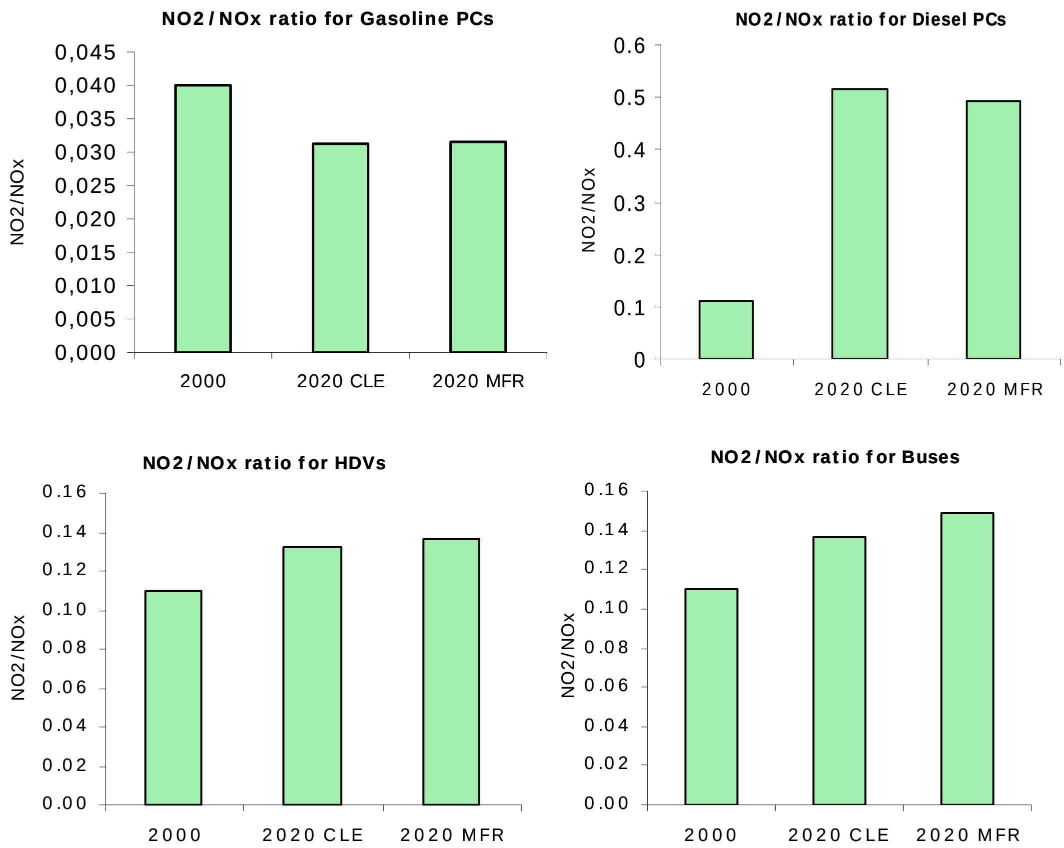




However, one issue is the foreseen significant increase especially in NO_2 emissions for diesel passenger cars, compared to the 2000 levels. This appears since it is expected that all passenger cars will have high performance oxidative exhaust aftertreatment installed, which significantly promotes the oxidation of NO to NO_2. This issue may have significant implications in the attainment of future air quality standards.

For HDVs and buses there is a lower NO_2/NO_x ratio expected than for passenger cars, due to the earlier introduction of SCR systems (in some cases starting already from Euro IV) and the longer lifetime of older HDVs with low NO_2 emissions. SCR leads to a low NO_2/NO_x ratio. On the other hand, the use of CRT DPFs increases the NO_2/NO_x ratio. As a result, the exact proportion of NO_2 in vehicle exhaust is not easy to predict, since it depends on the sequence of SCR and CRT in the exhaust line. Figure 3.2 corresponds to the best estimate that can be attempted today.

Figure 3.2 Primary NO_2/NO_x ratio of different vehicle categories

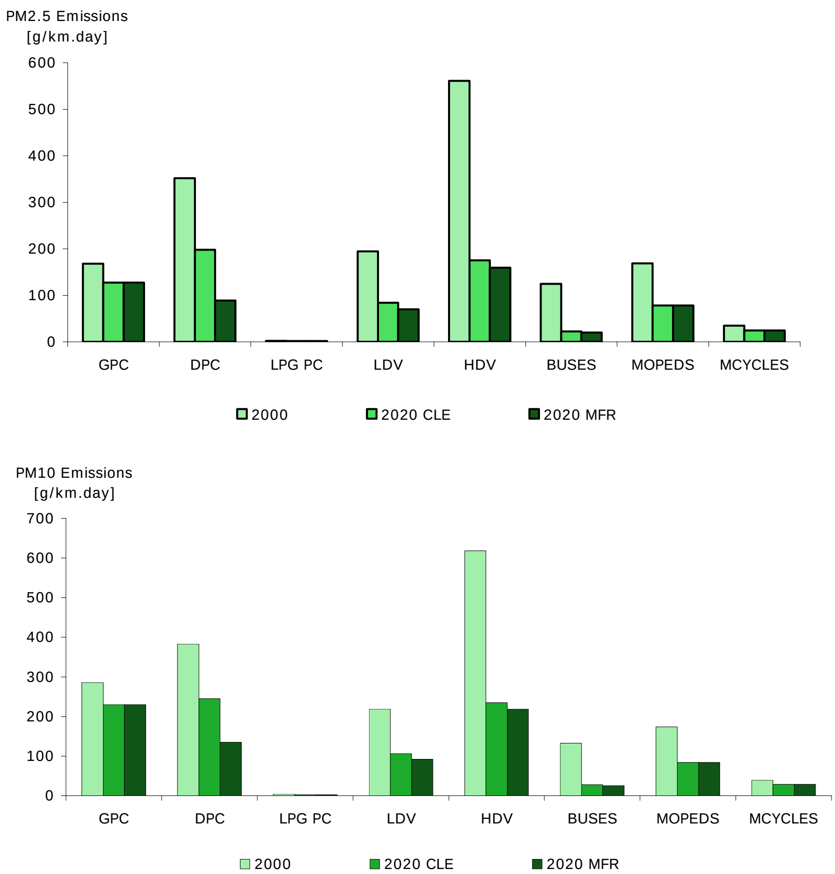


3.3 Total $PM_{2.5}$ and PM_{10} emissions per category

Primary PM emissions have been calculated including both exhaust and non-exhaust emissions, except re-suspension. The contribution of non-exhaust particles is more prominent in the coarse mode (2.5-10 μm). In 2000, the main contributor in both particle size ranges is HDVs. This is despite the share of vehicles >20 t, which has been set to zero for this urban network (see section 2.1.2.2). In 2000, the second most important contributor of $PM_{2.5}$ is diesel passenger cars. This is because of the high exhaust PM emission factor of such vehicles compared to gasoline ones. Mopeds are also significant emitters, in particular of $PM_{2.5}$, in this urban network. This is due to the relatively high share of 2-stroke motorcycles with high PM exhaust emission rates.

The situation will largely change in the future (both 2020 CLE and in particular the 2020 MFR) scenarios, as diesel vehicles are equipped with DPFs. In this case vehicle number is more important than the fuel used, due to the reduction in exhaust emissions from diesel cars. As a result, total PM emissions of gasoline passenger cars will exceed the diesel ones in the future due to their higher numbers and the fact that their non-exhaust emission factors do not differ from diesel vehicles. In fact, gasoline passenger cars as a whole appear as the most significant PM_{10} contributor for the 2020 MFR scenario. This shows that although the various exhaust PM control measures can be effective they will fail to lead to the attainment of very strict air quality standards due to the high contribution of non-exhaust emissions. The effect would have been even larger, if re-suspension was also included in the calculations. This needs to be considered in the future.

Figure 3.3 $PM_{2.5}$ and PM_{10} emissions from different vehicle categories per km street canyon with 20.000 vehicles/day as a function of the scenario considered. Projections are based on expected emission factor reductions for new technologies and changes in the vehicle stock mix

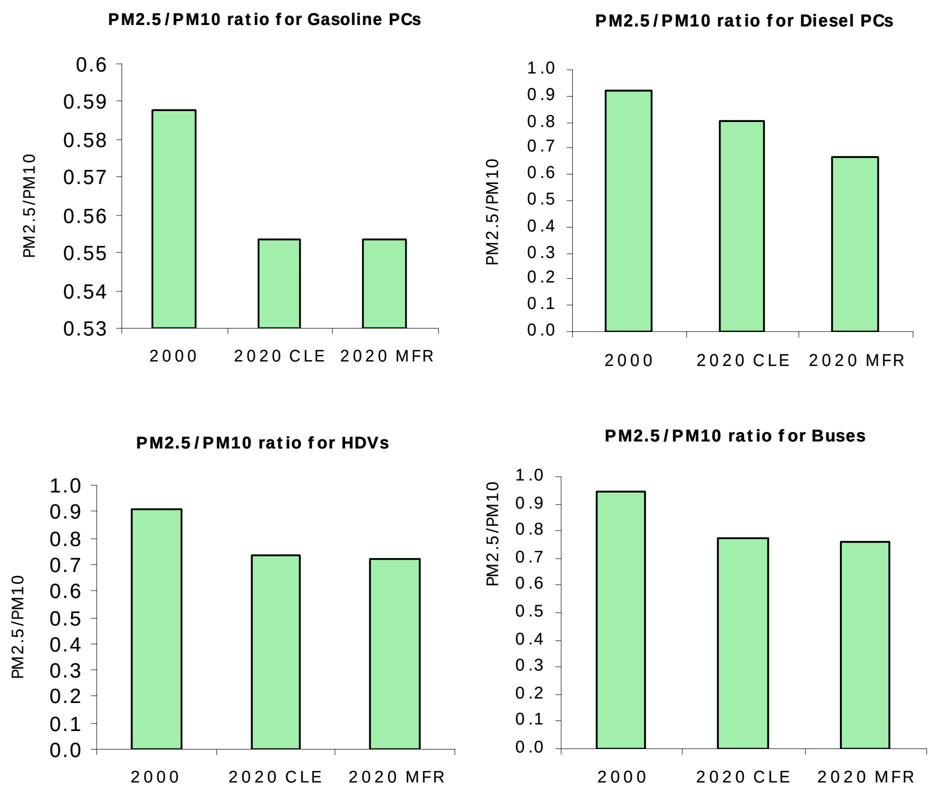


3.4 Evolution of the $PM_{2.5}/PM_{10}$ emissions ratio

The ratio of $PM_{2.5}/PM_{10}$ emission factors as a function of the scenario and vehicle category considered is shown in Figure 3.4. The ratio changes for future scenarios, as the contribution of exhaust vs. non-exhaust PM decreases. The gasoline passenger car ratio is the lowest, as exhaust PM emissions are very low already since 2000. This further reduces in the future as new gasoline technologies have even better combustion control, hence lower $PM_{2.5}$ emissions, than older vehicles.

Diesel vehicle PM emissions in 2000 are practically dominated by $PM_{2.5}$. Over 90% of total $PM_{2.5}$ is exhaust emissions, because only a very limited number of vehicles are equipped with DPFs. This changes in the future as exhaust emissions decrease, while non-exhaust emissions remain practically constant. This is also important for the prediction of $PM_{2.5}$ vs. PM_{10} attainment in the future.

Figure 3.4 $PM_{2.5}$ and PM_{10} emissions from different vehicle categories as a function of the scenario considered (Note different y-axis of Gasoline PCs compared to other categories)



3.5 Diurnal distribution of emissions

The diurnal distribution of the NO_x emissions for all cities for the year 2000 (Figure 3.5) shows that NO_x emissions approach the highest values from 17:00 to 19:00, that is during the evening rush hour. Gasoline passenger cars and HDVs have the highest contribution to NO_x emissions. The same diurnal profile also appears for NO_2 emissions. However, the absolute levels are significantly lower.

The emissions of PM follow the same diurnal profile but with a different relative contribution of the various vehicle categories, compared to NO_x emissions. In this case, $PM_{2.5}$ and PM_{10} levels do not differ significantly because of the dominance of exhaust (mainly particles smaller than 2.5 µm) in total PM emissions.

Figure 3.5 to Figure 3.10 show the diurnal profiles of NO_x, NO_2, $PM_{2.5}$ and PM_{10} emissions for the 2020 CLE and 2020 MFR scenarios. The main change compared to the 2000 concerns the absolute emission levels rather than the diurnal behaviour.

Figure 3.5 Diurnal distribution of NO_x and NO_2 emissions over the reference year (2000), distinguished per vehicle category

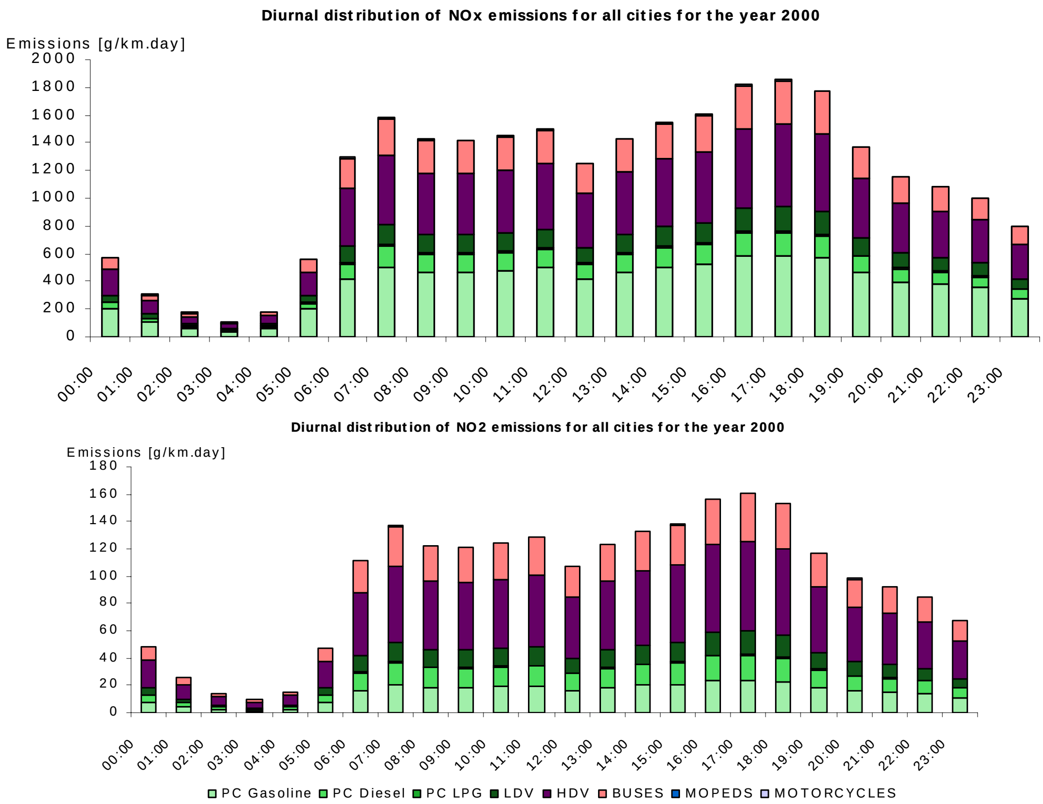
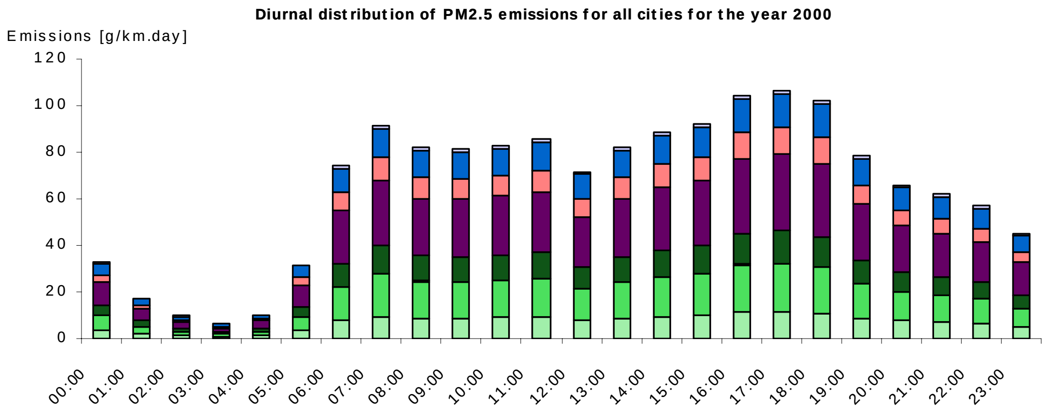


Figure 3.6 Diurnal distribution of $PM_{2.5}$ and PM_{10} emissions over the reference year (2000), distinguished per vehicle category



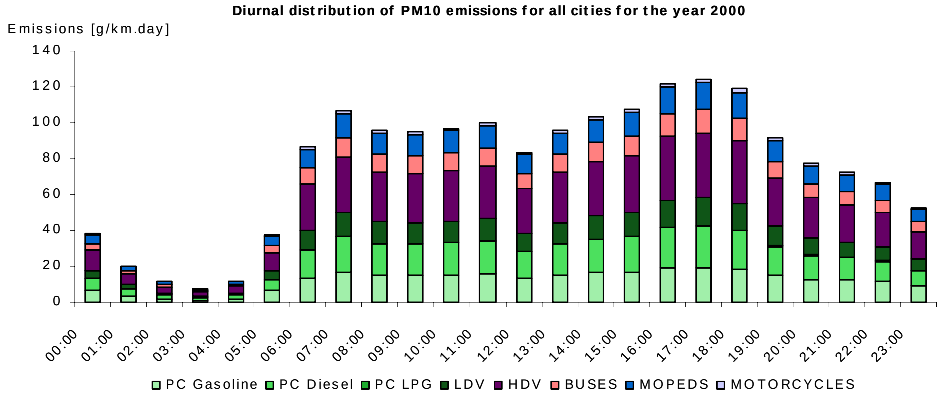


Figure 3.7 **Diurnal distributions of NO_x and NO_2 emissions over the 2020 CLE scenario, distinguished per vehicle category**

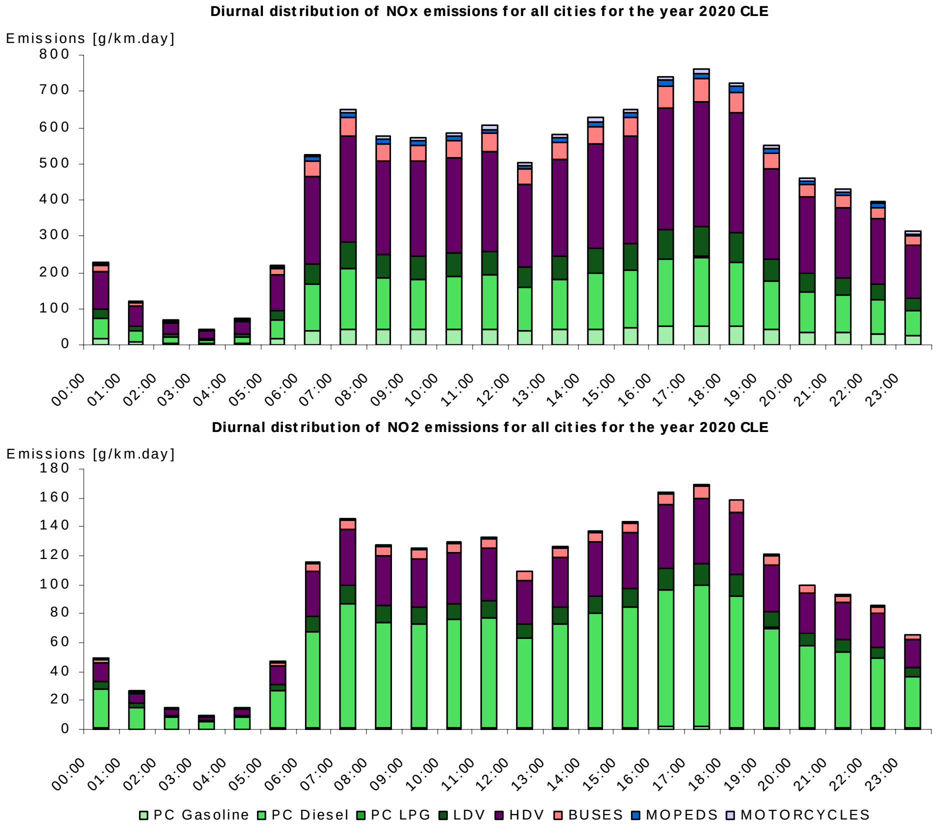


Figure 3.8 **Diurnal distributions of $PM_{2.5}$ and PM_{10} emissions over the 2020 CLE scenario, distinguished per vehicle category**

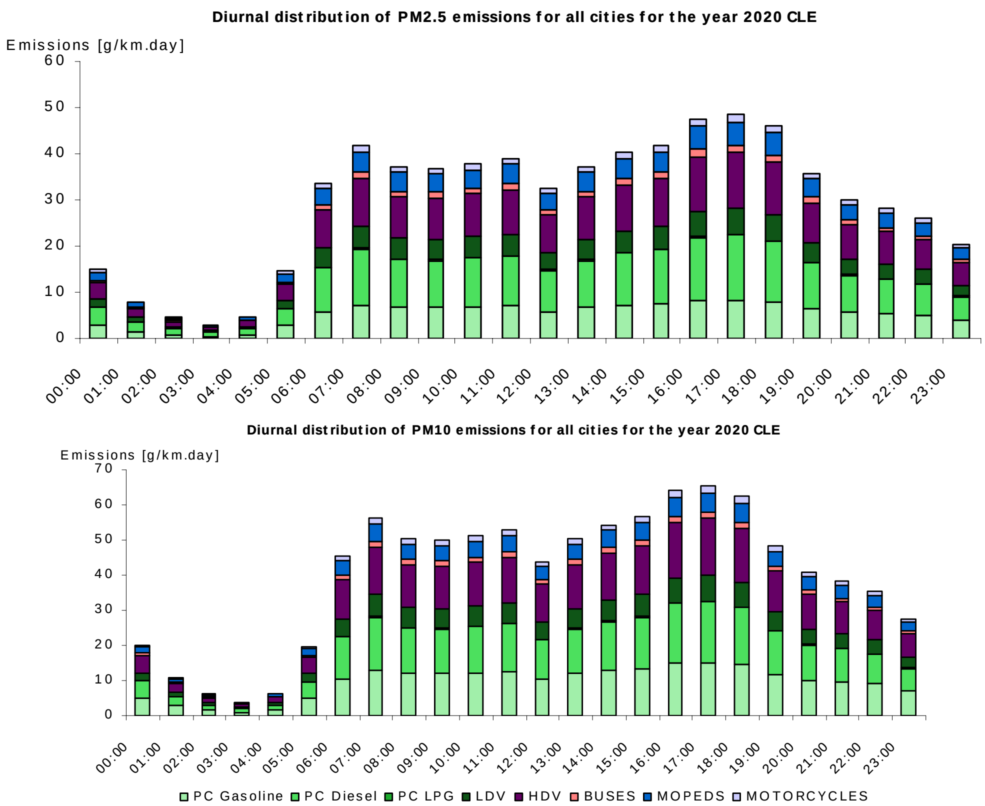


Figure 3.9 Diurnal distributions of NO_x and NO_2 emissions over the 2020 MFR scenario, distinguished per vehicle category

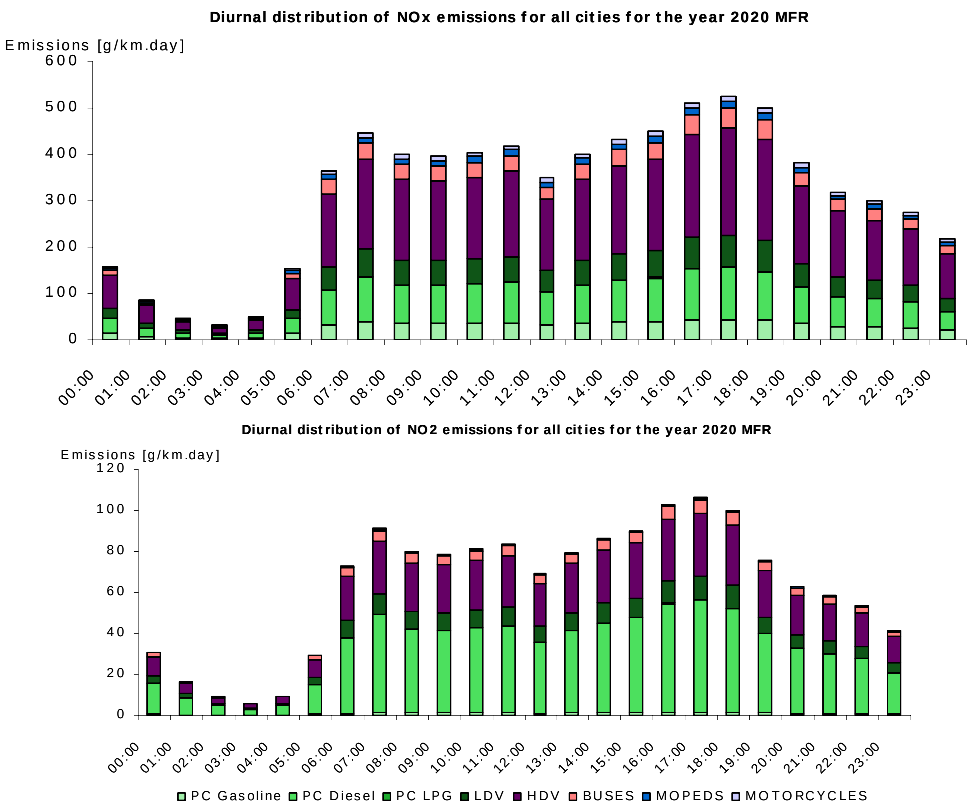
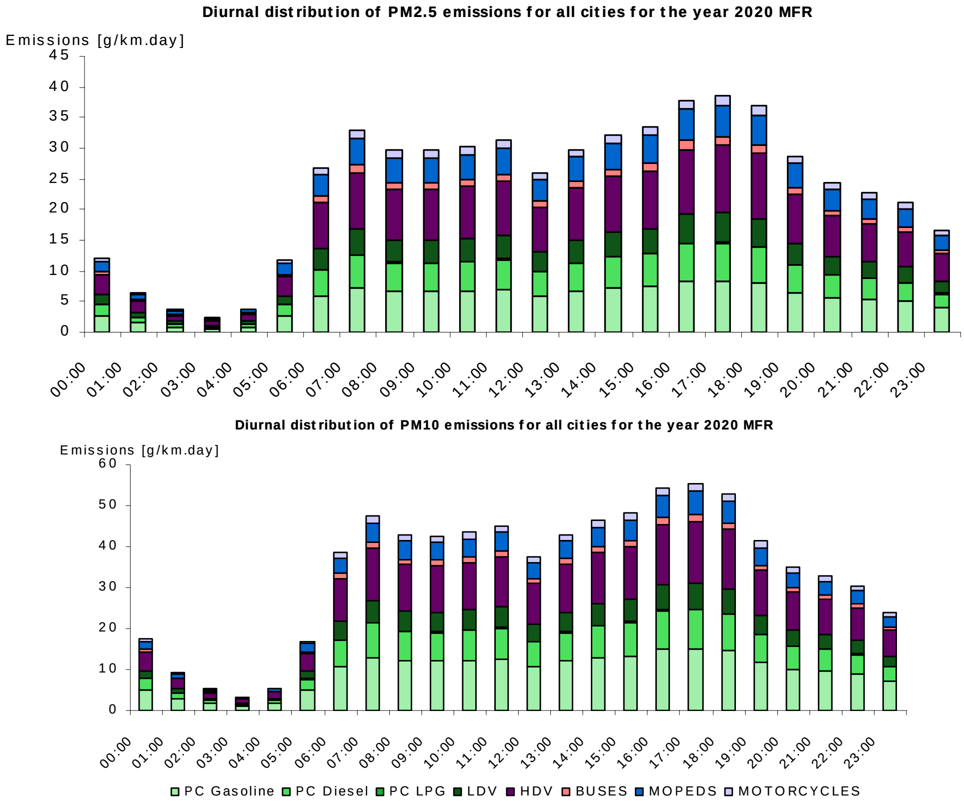


Figure 3.10 Diurnal distributions of $PM_{2.5}$ and PM_{10} emissions over the 2020 MFR scenario, distinguished per vehicle category



4 Local scale contribution to air pollution

In this chapter, the future air quality at urban and street level is investigated in terms of the annual mean concentrations of NO_2, NO_X, PM_{10} and $PM_{2.5}$ for the 20 cities, as well as hourly NO_2 and daily PM_{10} exceedances of the 2010 limit values, up for the year 2020. The current work is to a large extent complementary to the work which was performed within EEA's Technical Report No 1/2006 in the sense that an attempt is made to predict the evolution of the air quality across Europe up the year 2020, including also non EU-15 countries. Although particular attention is given to the local scale contribution to the air pollution levels at urban hotspots, results for the contribution of the urban and regional scales are also presented.

4.1 Street increments for 2020

Results regarding the projected street increments for 2020 for NO_2, NO_X, PM_{10} and $PM_{2.5}$ for the CLE and MFR scenarios for the 20 cities are presented in figures 4.1 – 4.4. In all cases and for all pollutants, compared to the reference year 2000, a reduced street increment is projected according to both the CLE and the MFR scenarios.

In particular, depending on the city:

- The modelled NO_2 street increment in 2000 ranged from 16–53 μg/m^3 while in 2020 between 12–32 μg/m^3 for the CLE and 7–24 μg/m^3 for the MFR scenario
- The modelled NO_x, street increment in 2000 ranged from 87–154 μg/m^3 while in 2020 between 27–56 μg/m^3 for the CLE and 12.2–45.2 μg/m^3 for the MFR scenario
- The modelled PM_{10}, street increment in 2000 ranged from 5–15 μg/m^3 while in 2020 between 2.3–5.2 μg/m^3 for the CLE and 1.9–4.8 μg/m^3 for the MFR scenario
- The modelled $PM_{2.5}$ street increment in 2000 ranged from 4–10 μg/m^3 while in 2020 between 1.5–4 μg/m^3 for the CLE and 1.1–3.6 μg/m^3 for the MFR scenario

According to the CAFE Scenario Analysis Report Nr. 2 (2004), significant reductions in the country scale emissions and hence in the corresponding urban scale emissions are foreseen for both scenarios. Indicatively, compared to the reference year 2000, the two scenarios assume average country scale reductions of:

- Up to 50% for NO_x and 45% for $PM_{2.5}$ for the CLE scenario
- Up to 70% for NO_x and 60% for $PM_{2.5}$ for the MFR scenario

Additionally, significant traffic emissions reductions are foreseen due to the introduction of new technologies assumed by both scenarios under consideration. For both CLE and MFR scenarios, the concentrations at street level have been calculated with the OSPM model, considering in each scenario (a) traffic emissions estimates for 2020 and (b) urban background concentrations calculated with OFIS (using urban scale emissions and thus considering reductions in all sectors). It is therefore ensured that the street level air pollution estimates are in line with the assumptions of the CLE and MFR scenarios, and corresponding emission reductions, made at both the local and urban scale.

Figure 4.1 Mean annual NO_2 street increment ($\mu g/m^3$) for 20 cities across Europe including four non EU-15 cities in 2000 compared to the projected street increment in 2020

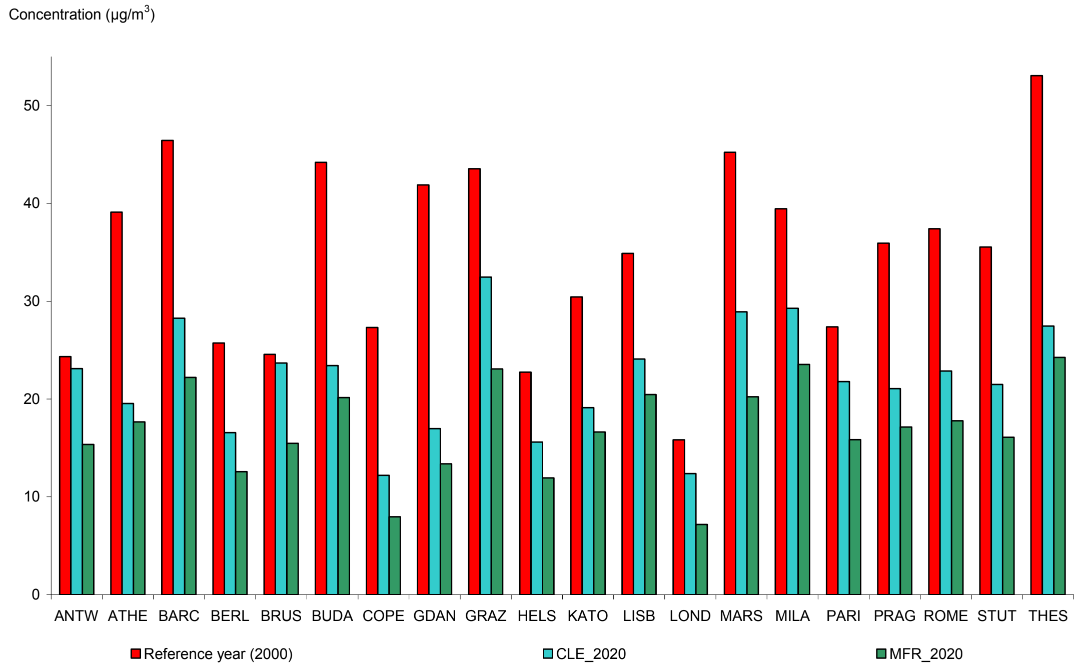


Figure 4.2 Mean annual NO_x street increment ($\mu g/m^3$) for 20 cities across Europe including four non EU-15 cities in 2000 compared to the projected street increment in 2020

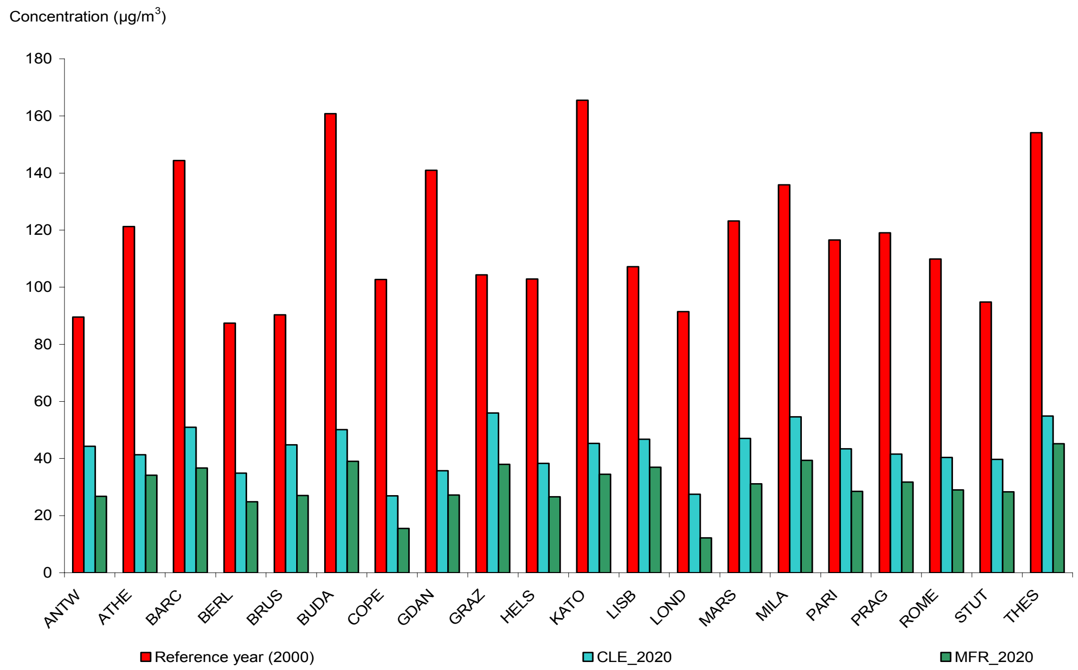


Figure 4.3 Mean annual PM_{10} street increment ($\mu g/m^3$) for 20 cities across Europe including four non EU-15 cities in 2000 compared to the projected street increment in 2020

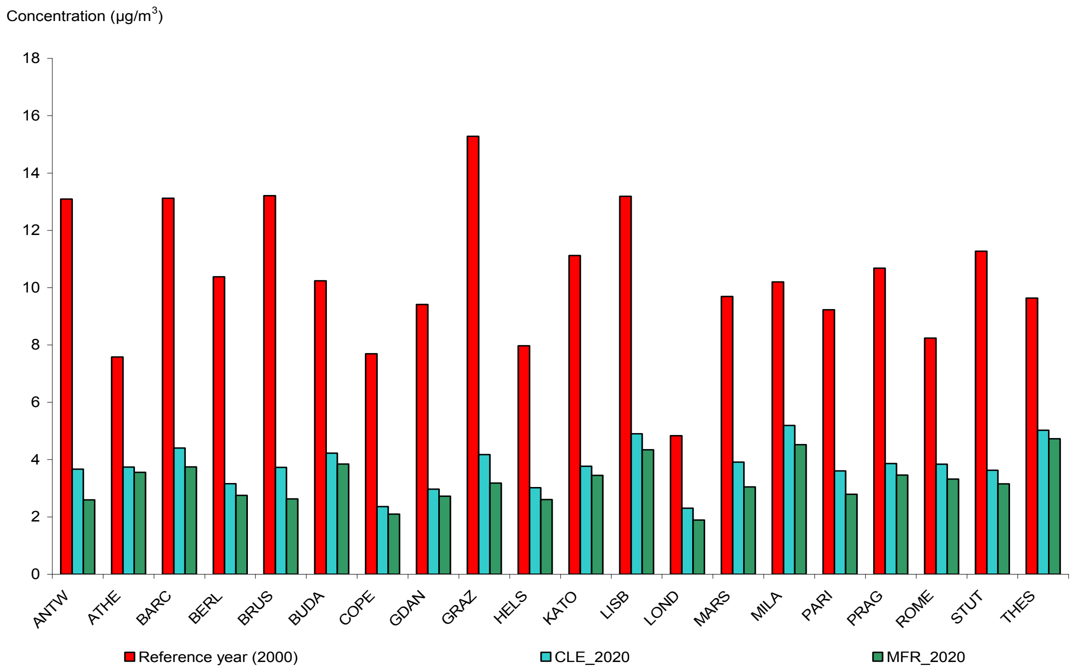
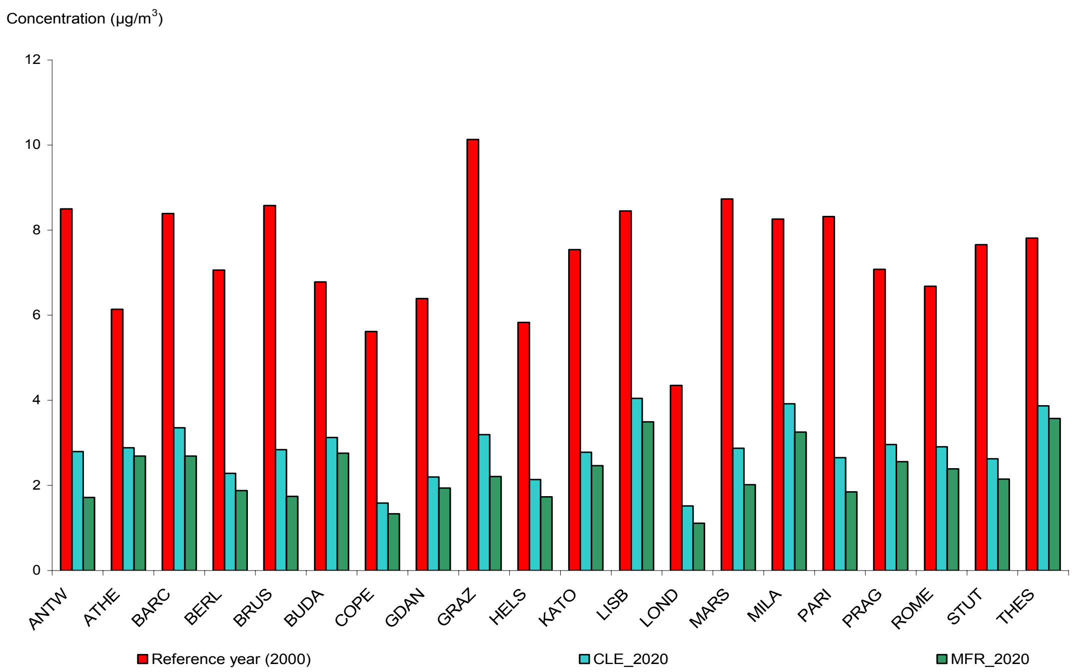


Figure 4.4 Mean annual $PM_{2.5}$ street increment ($\mu g/m^3$) for 20 cities across Europe including four non EU-15 cities in 2000 compared to the projected street increment in 2020



In order to gain a perspective of the evolution of the overall air pollution levels up to 2020, comparison of the individual contribution of the regional (EMEP model), urban (OFIS model) and the local (OSPM model) scales to the annual mean air pollution levels for NO_2 and PM_{10} across Europe for both scenarios has also been included. These results are presented in figures 4.5 and 4.6 for a number of cities, for NO_2 and PM_{10} respectively.

Figure 4.5 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 and the CLE and MFR scenarios for the year 2020

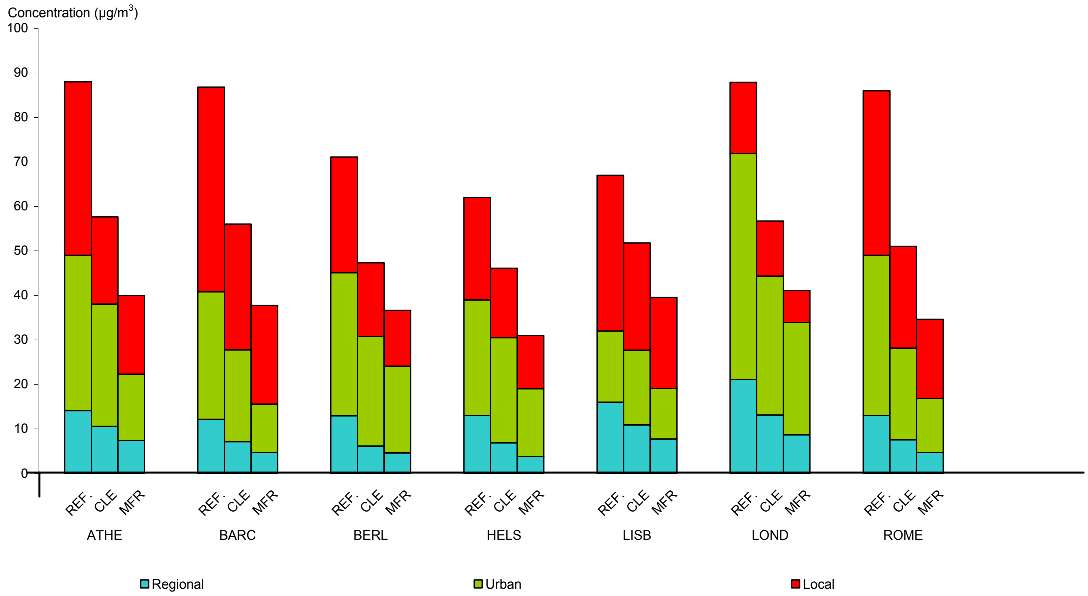
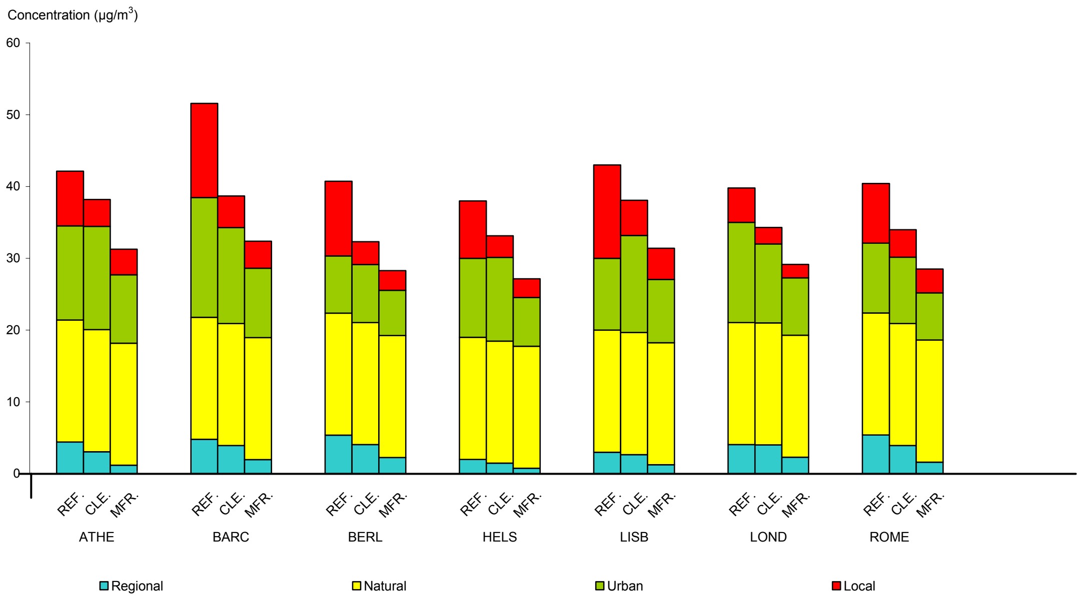


Figure 4.6 PM$_{10}$ annual mean air quality at regional scale (EMEP), urban scale (OFIS) and street scale (OSPM) for cities across Europe in the reference year and the CLE and MFR scenarios for the year 2020



It should be noted, that in line with the assumptions made during the 2006 study, OFIS results for PM$_{10}$, comprise a constant value of 17 µg/m^3 referred to as 'Natural' contribution, in order to account for natural PM sources such as windblown dust, sea salt and organic aerosols. The reason is that neither the regional (EMEP) nor the urban scale (OFIS) model accounts for natural primary PM sources and therefore a constant value 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) (EMEP, 2007). However, these stations are unevenly located across Europe since there are many countries with no data and as a result in some cases, the use of a constant value may lead to either an overestimation or an underestimation of the natural sources. Also, OFIS, like many urban scale models, does not yet account for the formation of secondary organic particulates which forms an extra source of uncertainty that could lead to a small underestimation of the modelled PM_{10} concentrations.

4.2 Hourly and daily exceedances in 2020

In this section, the predicted number of the hourly exceedances of the 2010 limit value of 200 $\mu g/m^3$ for NO_2 and the daily exceedances of the 2005 limit value of 50 $\mu g/m^3$ for PM_{10}, for the 20 cities under consideration and for both scenarios are presented. Results are shown in figures 4.7 and 4.8. It should stressed out that the results presented in this chapter are only indicative and should not be seen as representative for the typical hot spot situation in each one of the selected cities. The reason is that the number of exceedances of a certain threshold is particularly sensitive to inherent uncertainties of the underlying time series such as the uncertainty of the model, the uncertainty in the regional background calculations and the uncertainty due to the difference between local reality and standardised street canyon assumptions.

For all cities under consideration, the reduction of the number of predicted exceedances both for NO_2 and PM_{10} are significant compared to the modelled exceedances in the reference year 2000, for both scenarios. Especially for NO_2, for both scenarios, the results predict no hourly exceedances of the 2010 limit value of 200 $\mu g/m^3$ for 2020 (not to be exceeded more than 18 times per year).

As regards PM_{10} although according to the results the number of exceedances for the CLE scenario is significantly reduced, the maximum allowed number of 7 days per year is still exceeded in twelve cities. In the MFR scenario, numerical results predict that most cities will have close to zero exceedances with the exception of Antwerp and Paris, which will still potentially violate the allowed number of exceedances of 7 days per year (24 days per year for both cities). Once again however, it should be noted that no safe conclusions can be drawn on the basis of these results, since the various assumptions that have been made for the needs of the study (e.g. orientation and specific characteristics for the hypothetical street canyons) may have led to an underestimation of the predicted exceedances (for further details see EEA (2006), Section 4.2.2).

Figure 4.7 Number of hourly NO_2 exceedances of the 200 $\mu g/m^3$ limit value in 20 cities across Europe in 2000 and the CLE and MFR scenarios for 2020

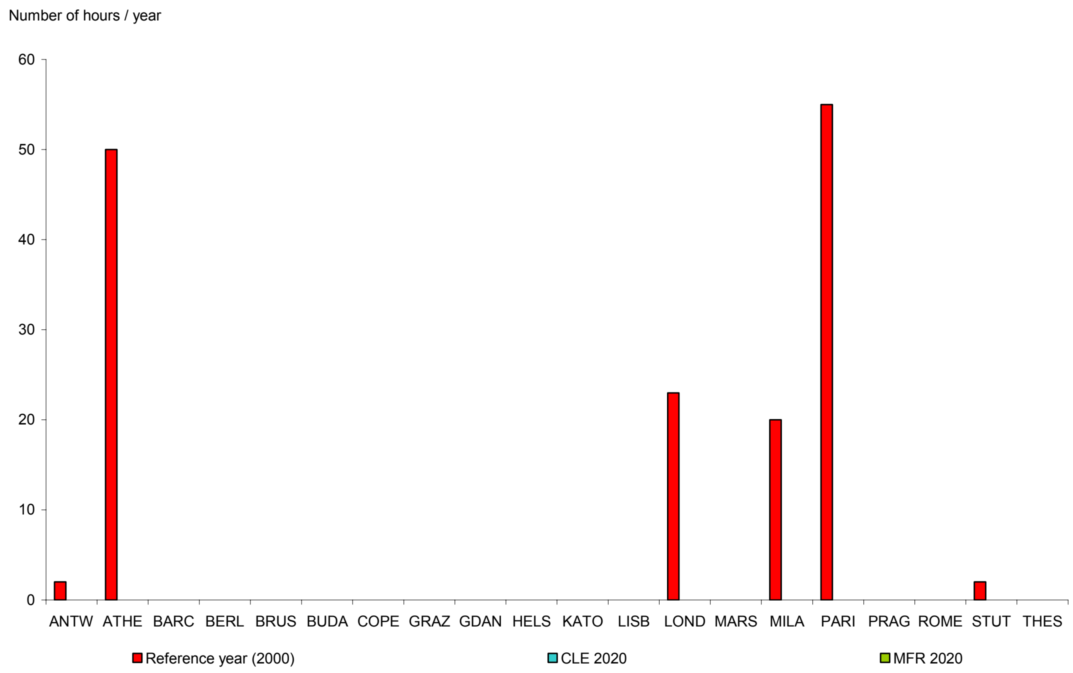
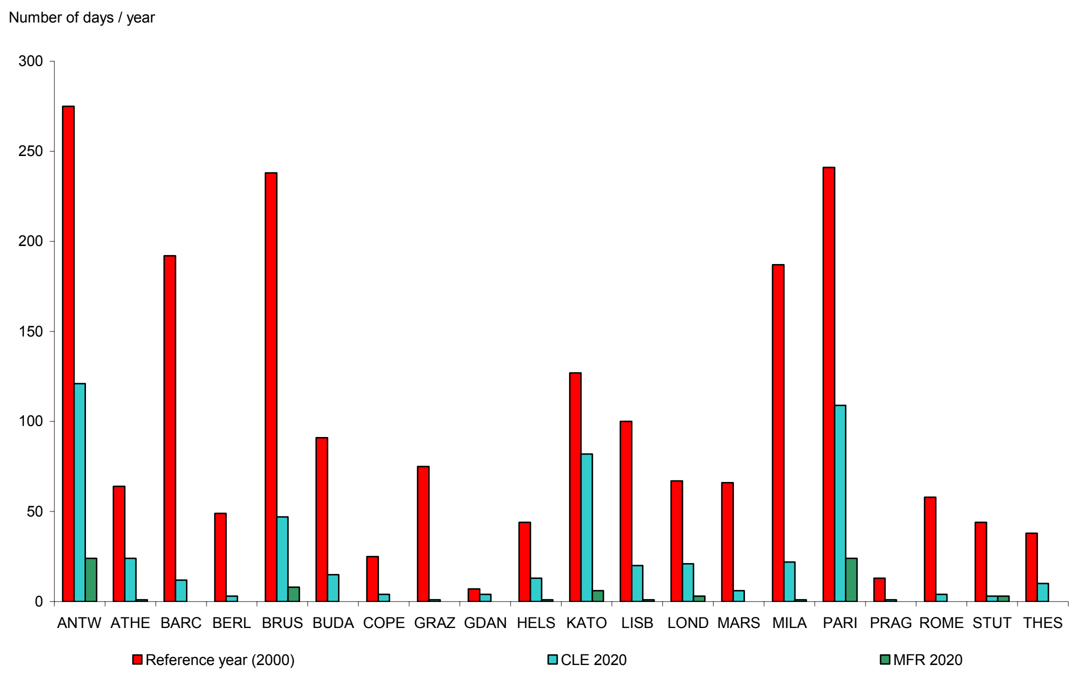


Figure 4.8 Number of daily PM_{10} exceedances of the 50 μg/m^3 limit value in 20 cities across Europe in 2000 and the CLE and MFR scenarios for 2020



It should be noted at this point that as regards the number of the estimated exceedances presented in this chapter there an underestimation. The reason is that in reality, there is significant contribution of re-suspension to non-exhaust emissions and the related air pollution levels at a canyon scale. However, re-suspension is not considered as a primary but rather as a secondary source of PM and hence its expected effect on the air quality at street scale has not been taken into account.

4.3 Retrofit scenario

In addition to the CAFE CLE and MFR scenarios, estimations for the air pollution levels at urban hotspots were made for three cities on the basis of the Retrofit scenario. The Retrofit scenario was assumed to correspond to an option where local (municipal or urban) authorities seek to do more than what emission standards require, in order to solve specific pollution problems (e.g. urban hotspots). In principle, it was assumed that a range of technologies will be retrofitted with advanced aftertreatment systems to examine what additional reductions can be achieved.

4.3.1 Street increments for the Retrofit scenario

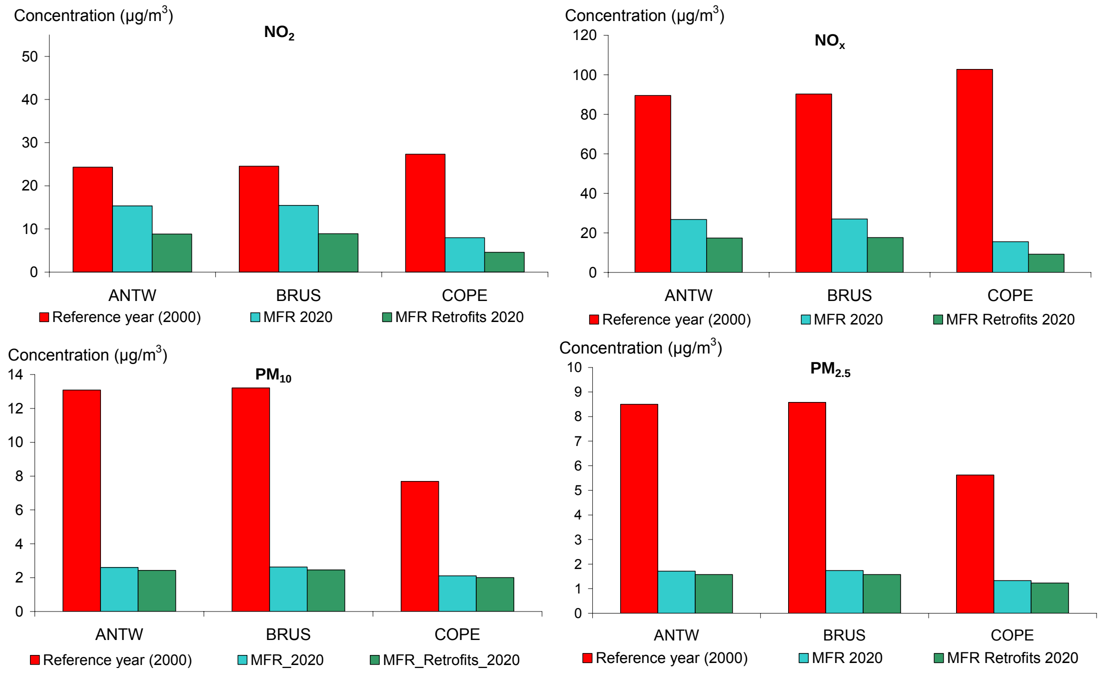
Since the retrofit scenario assumes the same traffic fleet configuration as the MFR scenario, comparison could only be made with MFR scenario. Two different options were examined one referring to NO_x (Selective Catalytic Reduction - SCR retrofits) and one referring to PM emissions (Diesel Particle Filters - DPF retrofits). The three cities were selected on the basis of the need to cover as many cases as possible and limitations in data availability. Both retrofit scenarios were applied for all three cities.

Results for the estimated street increments in all three cities and for all pollutants under consideration are presented in figure 4.9. Compared to the MFR scenario, significant reductions are shown both for NO_2 and NO_x. However, as regards both PM_{10} and $PM_{2.5}$ the achieved reductions are very small. More specifically:

- as regards NO_2 the reductions for all three cities are around 42%
- as regards NO_x the reductions range from a minimum of 34% for Antwerp and Brussels to a maximum of 40% for Copenhagen
- as regards PM_{10} the achieved reductions range from a minimum of 4.5% Copenhagen to a maximum of 6% for Antwerp and Brussels
- as regards $PM_{2.5}$ the achieved reductions range from a minimum of 7% Copenhagen to a maximum of 10% for Antwerp and Brussels

These findings are in line with various traffic related emissions reductions technologies assumed by the MFR scenario. As regards both $PM_{2.5}$ and PM_{10}, the MFR scenario assumes that most of the cars will be retrofitted with advanced aftertreatment systems anyway. Therefore, the proposed additional aftertreament systems in the Retrofit scenario (DPF) had only a small effect on the estimated PM street increment. The MFR scenario assumes that any advanced after treatment system necessary to comply with the required 80% NO_x reduction compared to Euro V will be implemented to Euro 6 light duty and Euro VI heavy duty vehicles. On the other had, the Retrofit scenarios, which were executed using the MFR fleet classification, assumes that advanced aftertreatment devices are applied to older technology vehicles (e.g. Euro 3) through retrofits. As a result, compared to the MFR scenario, the proposed system in the Retrofit scenario (SCR) resulted in significant reductions both for NO_2 and NO_x.

Figure 4.9 Mean annual street increment (µg/m^3) for the cities of Antwerp, Brussels and Copenhagen in 2000 compared to the projected street increment in 2020, for the MFR and Retrofits scenario



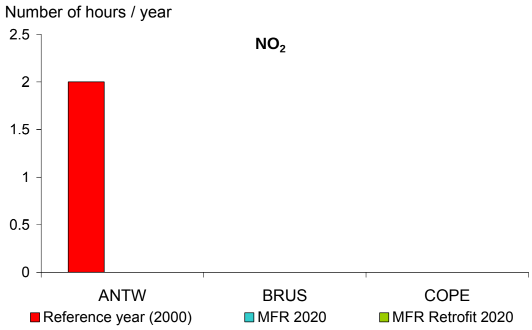
4.3.2 Exceedances for the Retrofit scenario

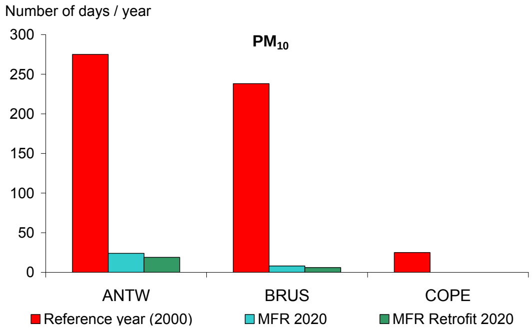
Results for the estimated expected exceedances for NO_2 and PM_{10} for Retrofit and the MFR scenarios for the year 2020 are shown in figure 4.10. For all three cities under consideration, the effect of the proposed aftertreatment systems of the Retrofit scenario have only a small impact on the number of predicted exceedances both for NO_2 and PM_{10}. As regards NO_2, results for the MFR scenario shown no hourly exceedances of the 2010 limit value of 200 µg/m^3 for 2020 (not to be exceeded more than 18 times per year) in the first place. As regards PM_{10}, the estimated daily exceedances of the 50 µg/m^3 limit value in the cities of Antwerp and Brussels are only slightly reduced. In Brussels, the total number of daily exceedances drops from 8 to 6 and as a result the maximum allowed number of daily exceedances (7 days per year) is not violated. However, in Antwerp the maximum allowed number of daily exceedances per year is still violated, since the predicted number of daily exceedances is 19.

It should be noted, that in the case that retrofit schemes are implemented over larger spatial scales, compared to small areas in cities with just a few street sections, urban background levels will also fall. As a result, the benefits in terms of reduced number of exceedances that can be attributed to the implementation of retrofit schemes may be underestimated. Furthermore, the results for the number of exceedances in the selected three cities should not be considered as representative for other cities since the effect in terms of reduced number of exceedances strongly depends on how much the initial levels exceed the threshold. As a result, the estimated number of exceedances is prone to high uncertainties, since they depend mainly on the absolute total concentrations levels.

Figure 4.10 Number of NO_2 hourly exceedances of the 200 µg/m^3 limit value and daily PM_{10} exceedances of the 50 µg/m^3 limit value in Antwerp, Brussels and Copenhagen in 2000 and the \MFR and Retrofit scenarios for 2020

Local scale contribution to air pollution





5 Health impacts assessment

Within the frame of this chapter, physical impacts on human receptors that are exposed to air pollution at both urban and street level in the 20 European cities under investigation are studied on the basis of CLE and MFR scenarios. In particular, loss of Life Expectancy (LE) for chronic mortality of the entire population and morbidity related to cases of chronic bronchitis for adults due to long-term exposure to PM_{10} and $PM_{2.5}$ are estimated. Assessment of the health impacts based on chronic mortality and chronic bronchitis is usually performed using selected Exposure-Response Functions[5] (ERFs) that correlate PM_{10} and $PM_{2.5}$ concentrations with the specific health endpoints. In the analysis that follows, for simplicity and without substantial loss of generality and in an effort to assess the overall health impact across all 20 European cities, a single ERF is adopted for every physical impact under consideration.

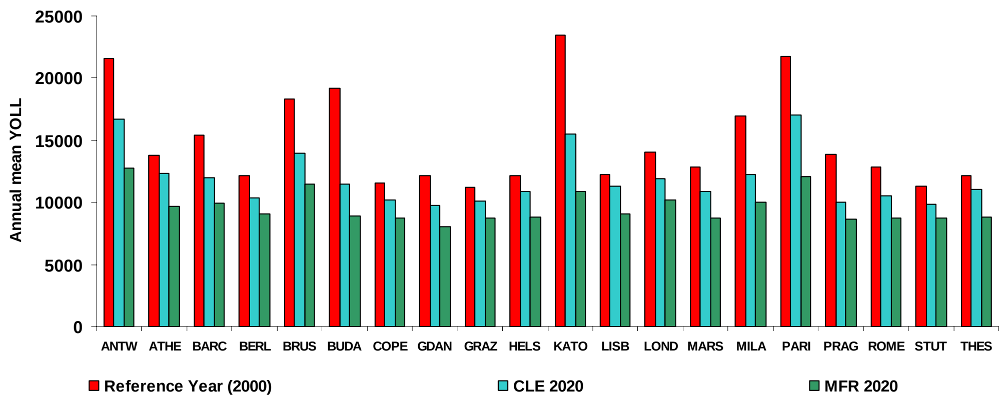
5.1 Analysis

Life Expectancy (LE) loss was estimated for every individual city for the reference year 2000 and for both scenarios for the target year 2020 in terms of Years Of Life Lost (YOLL) due to the long term exposure to PM_{10} and $PM_{2.5}$ concentrations. In addition, the number of chronic bronchitis cases was estimated also for every individual city under consideration for the reference year 2000 and for both scenarios fort he target year 2020.

5.1.1 Loss of life expectancy (LE) for chronic mortality

Figures 5.1 – 5.4 present the estimated loss of LE for chronic mortality for the 20 cities across Europe in 2000 compared to projected loss in 2020 for CLE and MFR. The estimated loss of LE is expressed in terms of the annual mean years of life lost (YOLL) per million inhabitants.

Figure 5.1 Long term exposure to PM_{10}. Annual mean YOLL due to chronic mortality in 2000 compared to the projected ones in 2020 per million inhabitants for 20 cities across Europe



[5] In addition the terms Concentration – Response functions (CRFs) or Dose-Response functions (DRFs) are often used. Although an ERF should be based on the dose actually absorbed by a receptor, it is often used in a wider sense where it is formulated directly in terms of the concentration of a pollutant in the ambient air, accounting implicitly for the absorption of the pollutant from the air into the body.

According to figure 5.1, Katowice, Paris, Antwerp, Budapest and Brussels are the cities with the largest loss in LE. For these cities, for the year 2000 the mean annual estimated YOLL was 23500 for Katowice, more than 21500 for Paris and Antwerp, approximately 19200 for Budapest and 18500 for Brussels. The cities which show a smaller loss of LE are Graz, Stuttgart and Copenhagen. Long term exposure to PM_{10} concentrations, averaged over the 20 cities under consideration contributes to 15000 YOLL for 2000, 11900 for 2020 for the CLE scenario and 9600 for 2020 for the MFR scenario. The total loss of LE for the 20 cities under consideration (as an indicator for 20 million receptors) is approximately 300000 YOLL for the reference year 2000, 238000 for 2020 CLE and 192000 for 2020 MFR. Indicatively, 300000 YOLL can be translated either to a total of 300000 premature deaths by 1 year, or to a total of 30000 premature deaths by 10 years.

Figure 5.2 presents the estimated annual mean YOLL due to traffic related PM_{10} concentrations at street level, for chronic mortality per million inhabitants for all cities and for both scenarios. This YOLL estimation accumulates on the top of the expected health outcomes due to long term exposure to PM_{10} at urban scale only for the inhabitants of the core of the cities which are exposed in the specific street increment inside street canyons and not in all population of the metropolitan areas.

Based on the results presented in figure 5.2 the cities with the larger loss of LE due to PM_{10} traffic related street level concentrations are Graz, Brussels, Barcelona, Antwerp and Lisbon. For the reference year 2000 the total estimated YOLL for Graz exceeded 6200 while for the cases of Brussels, Antwerp, and Barcelona it was estimated to approximately 5500, while in the case of Lisbon the corresponding YOLL was 5300. The city with the smallest amount of YOLL was London, with an estimated YOLL of approximately 2000. The total average YOLL across the 20 cities under consideration due to traffic-related long term exposure to PM_{10} at street level was estimated to 4300 YOLL for 2000, 1560 for 2020 for the CLE scenario and 1320 for 2020 for the MFR scenario. Furthermore, the total loss of LE as a sum of the YOLL over the 20 cities under consideration was approximately 85500 YOLL for 2000, 31000 for 2020 for the CLE scenario and 26500 for 2020 for the MFR scenario. The share with respect to the total amount of YOLL that the traffic related PM_{10} pollution is responsible for 22% of the total YOLL in 2000, 11.5% of the total YOLL in the year 2020 for the CLE scenario and 12% of the total YOLL in the year 2020 for the MFR scenario.

Figure 5.2 Long term exposure to traffic related PM_{10} concentrations at street level. Annual mean YOLL for chronic mortality for cities across Europe in 2000 compared to the projected ones in 2020 per million inhabitants

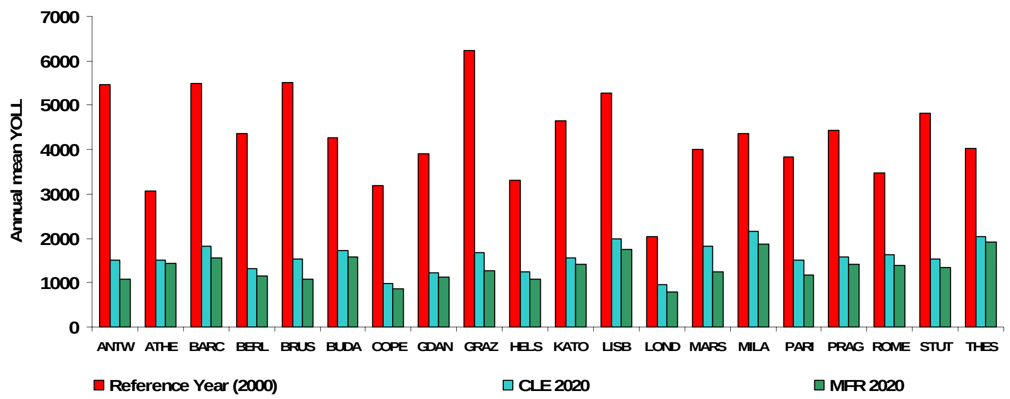


Figure 5.3 presents the expected YOLL due to long-term exposure to $PM_{2.5}$. Paris, Katowice, Antwerp followed by Brussels and Milan are the cities with the largest loss of LE in terms of YOLL. In particular, in Paris for the reference year 2000 there were an estimated 21500 YOLL, approximately 17000 for Katowice and Antwerp and 15500 for Brussels and Milan. The cities with the less expected loss of LE were Graz, Gdansk Stuttgart and Copenhagen. Long term exposure to $PM_{2.5}$ averaged over the 20 cities under consideration contributes to approximately 10800 YOLL for 2000, 6800 for 2020 CLE and 3600 for 2020 MFR. As an indication of the total YOLL across Europe, the total loss of LE for the 20 cities under consideration, is approximately 215000 YOLL for 2000, 135000 for 2020 CLE and 72000 for 2020 MFR.

Figure 5.4 shows the expected annual mean YOLL due to traffic related $PM_{2.5}$ street level concentrations based on estimates for chronic mortality per million inhabitants. For the reference year 2000, the cities with the largest loss of LE in terms of YOLL were Graz, Marseille, Brussels, Antwerp and Barcelona. In particular Graz suffered from an estimated 7000 YOLL, while the other four cities from approximately 6000 YOLL. On the other hand, London experienced the least loss of LE with an estimated 3000 YOLL. Long term street level exposure to traffic related $PM_{2.5}$ concentrations calculated as an average over the 20 cities in question, resulted in approximately 5100 YOLL for 2000, 1900 for the CLE and 1560 for the MFR scenarios for the target year 2020. The total loss of LE for the 20 cities under consideration is approximately 102000 YOLL for 2000, 38000 for the CLE and 31000 for MFR scenarios for the year 2020. In other words, traffic related pollution is responsible for 32% of the total YOLL in 2000, 22% of the total YOLL in the year 2020 for the CLE scenario and 30% of the total YOLL in the year 2020 for the MFR scenario. As regards the estimated YOLL due to chronic mortality, compared to the YOLL that can attributed to the traffic related PM_{10} concentrations, the corresponding YOLL due to the traffic related $PM_{2.5}$ concentrations is slightly higher. This is the direct result of the specific response functions that were selected for the needs of the study. More specifically, the correlation coefficient used in the response function which relates $PM_{2.5}$ concentration with various heath endpoints, is 40% higher than the corresponding one used for PM_{10}. At the same time, for the CLE and MFR scenarios the average PM_{10} concentrations are 26% and 30% higher than the average $PM_{2.5}$ concentrations respectively. Therefore, at all times $R_{CM,PM_{2.5}} > R_{CM,PM_{10}}$ (see Chapter 2.3). In effect, this means that the negative impact of exposure to $PM_{2.5}$ concentrations on human health is more intense compared to the impact due to exposure on PM_{10} concentrations.

Figure 5.3 Long term exposure to $PM_{2.5}$. Annual mean YOLL for chronic mortality for cities across Europe in 2000 compared to the projected ones in 2020 per million inhabitants

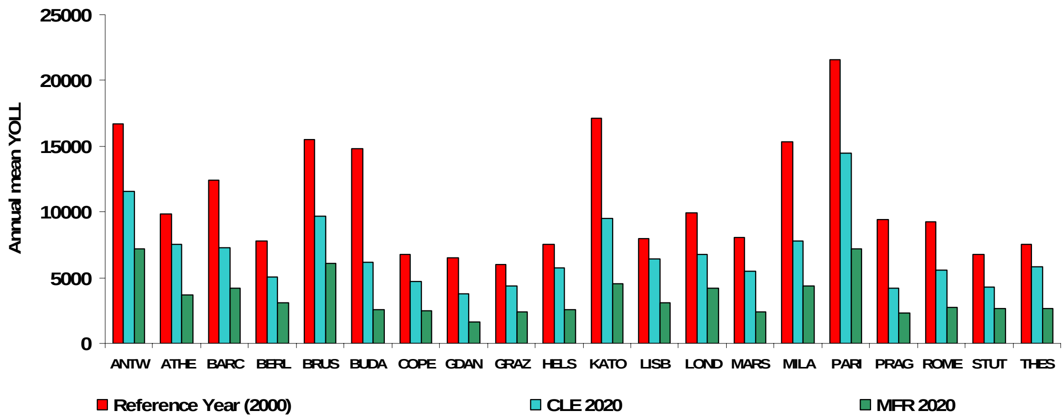
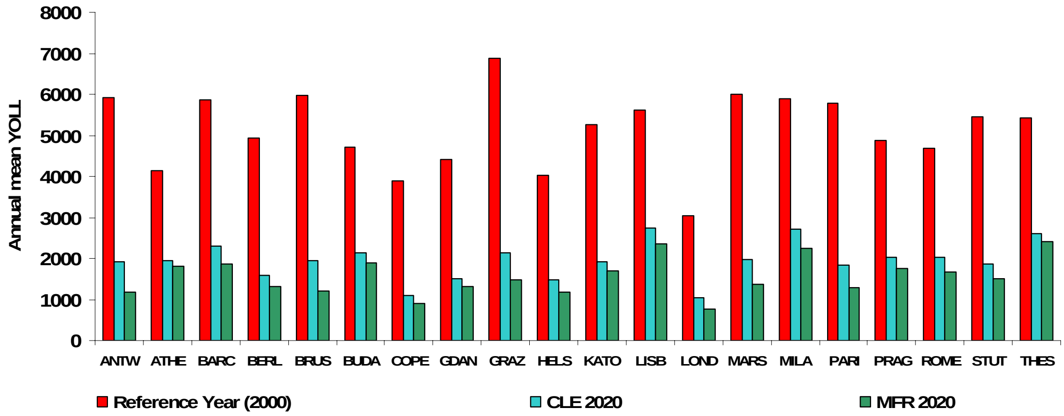


Figure 5.4 Long term exposure to traffic related $PM_{2.5}$ concentrations at street level. Annual mean YOLL for chronic mortality for cities across Europe in 2000 compared to the projected ones in 2020 per million inhabitants



5.1.2 *Cases of chronic bronchitis*

Similar to the loss of LE in terms of YOLL for chronic mortality, figures 5.5 – 5.8 present the estimated annual number of cases of chronic bronchitis for the 20 cities across Europe in 2000 compared to projected loss in 2020 for CLE and MFR scenarios. Katowice, Paris, Antwerp, Budapest and Brussels are the cities with the largest number of cases attributed to chronic bronchitis. In the cases of Katowice, Paris and Antwerp, the estimated number of attributed cases is approximately 1500 (figure 5.5). The cities with the least expected cases of chronic bronchitis are Graz, Stuttgart and Copenhagen. These results are in line with the estimated loss of LE in terms of YOLL due to chronic mortality, presented in figure 5.1. The average calculated cases of chronic bronchitis due to the long term exposure to PM_{10} for the 20 cities under consideration, were 1000 cases for 2000, 790 cases for 2020 for the CLE scenario and 640 for the MFR scenario for the target year 2020. The sum of the estimated chronic bronchitis cases for the 20 cities was approximately 19800 cases for 2000, 15700 for the CLE and 12700 for the MFR scenario for the year 2020.

Figure 5.5 Long term exposure to PM_{10}. Annual mean number of estimated cases of chronic bronchitis for cities across Europe in 2000 compared to the projected ones in 2020, per million inhabitants.

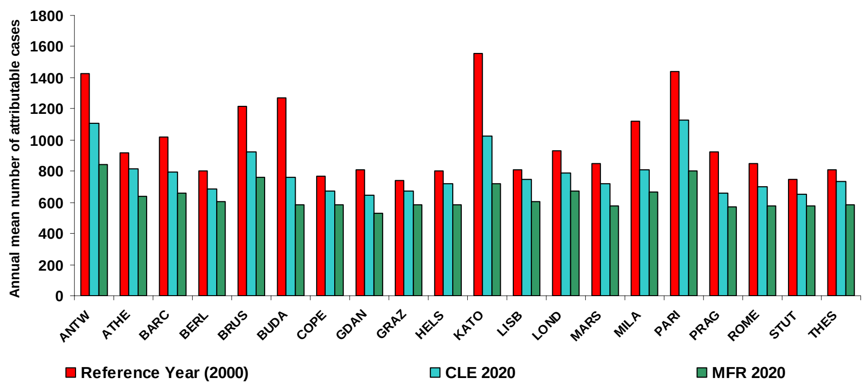
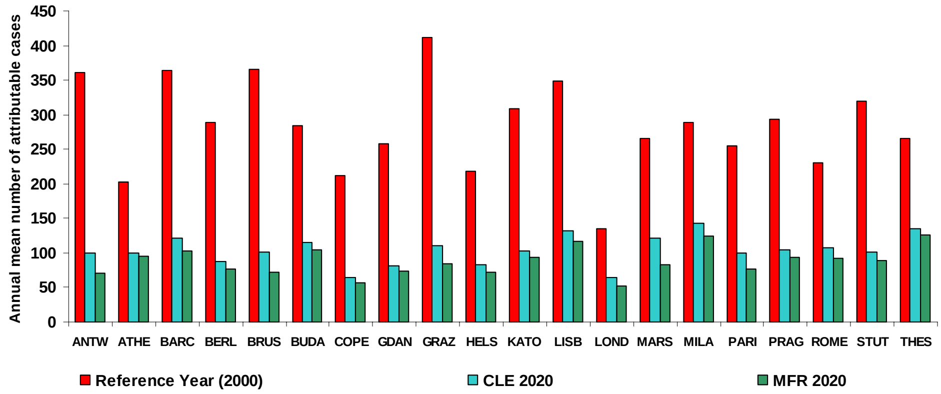


Figure 5.6 shows the annual mean number of cases to chronic bronchitis due to the local scale contribution to the PM_{10} concentrations at the street level. The contribution of the annual average

street increment to the number of chronic bronchitis cases is quite important, since it adds on the expected chronic bronchitis cases due to the long term exposure to PM_{10} at urban scale. However, since the street increment for all cities and for both scenarios was calculated assuming a street located in the centre of the cities, these results apply only to the inhabitants of the core of the cities.

Figure 5.6 Long term exposure to traffic related PM_{10}concentrations at street level. Annual mean number of estimated cases of chronic bronchitis for cities across Europe in 2000 compared to the projected ones in 2020 per million inhabitants.



The largest number of chronic bronchitis cases due to traffic related PM_{10} concentrations is estimated for the cities of Graz, Brussels, Antwerp, Barcelona and Lisbon. In the case of Graz For the reference year 2000 the estimated number of chronic bronchitis cases is approximately 400 for Graz and 360 for Brussels, Antwerp and Barcelona. The city with the least expected cases of chronic bronchitis is London with an estimated annual average of 135 cases. In an average, long term exposure to traffic related PM_{10} concentrations for the 20 cities under consideration results in 280 cases of chronic bronchitis for 2000, 105 cases for the CLE and 88 cases for the MFR scenario for the year 2020. Additionally, the total estimated number of chronic bronchitis cases for the 20 cities under consideration is approximately 5700 cases for 2000, 2070 for the CLE and 1750 for the MFR scenario for the target year 2020.

Figure 5.7 shows the estimated number of cases of chronic bronchitis due to the long term exposure to $PM_{2.5}$ concentrations at urban scale. The largest number of chronic bronchitis cases due to long term exposure to $PM_{2.5}$ concentrations was estimated for the cities of Paris, Katowice, Antwerp, Brussels and Milan. For the reference year 2000 the estimated number of chronic bronchitis cases is approximately 1450 for Paris and 1100 for Katowice, Antwerp, Brussels and Milan. The city with the least expected cases of chronic bronchitis due to the long term exposure to PM2.5 was Graz with an estimated annual average of 400 cases. In an average, long term exposure to $PM_{2.5}$ concentrations for the 20 cities under consideration resulted in 720 cases of chronic bronchitis for 2000, 450 cases for the CLE and 240 cases for the MFR scenario for the year 2020. Additionally, the total estimated number of chronic bronchitis cases for the 20 cities under consideration is approximately 14400 cases for 2000, 9000 for the CLE and 4750 for the MFR scenario for the target year 2020.

Figure 5.7 Long term exposure to $PM_{2.5}$. Annual mean number of attributable cases to chronic bronchitis of adults for cities across Europe in 2000 compared to the projected ones in 2020 per million inhabitants.

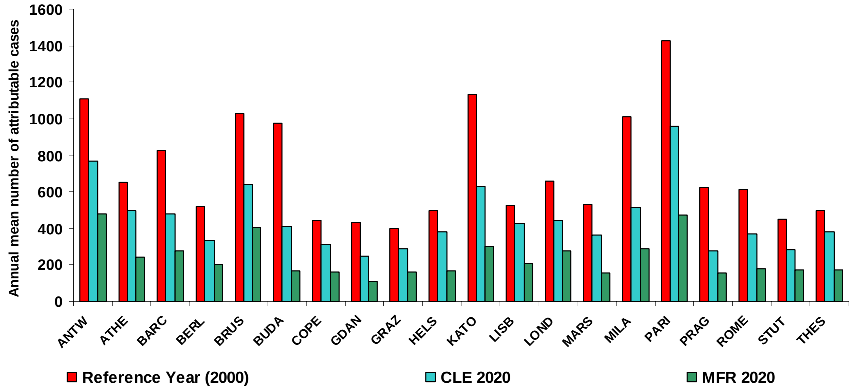
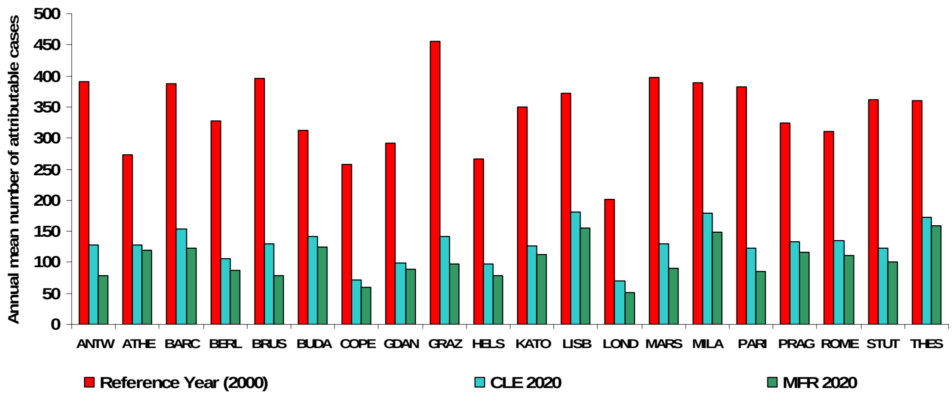


Figure 5.8 presents the number of estimated chronic bronchitis cases due to traffic related $PM_{2.5}$ concentrations at street level, per million inhabitants for both scenarios, for all cities. As regards the year 2000, the cities with the largest number of chronic bronchitis cases are Graz, Marseille, Brussels, Antwerp and Barcelona. In Graz the total number of cases attributed to chronic bronchitis is approximately 460, while in the other 4 cities it is 400 cases. The city with the least number of chronic bronchitis cases is London with an estimated 200. At average, for the each of the 20 cities under consideration, long term exposure to traffic related $PM_{2.5}$ concentrations at street level resulted in approximately 340 cases for 2000, 130 cases for 2020 CLE and 100 cases for 2020 MFR. The sum of the estimated cases for the 20 cities was approximately 6800 cases for 2000, 2570 for the CLE and 2070 for the MFR scenario for the target year 2020.

Figure 5.8 Long term exposure to traffic related $PM_{2.5}$ concentrations street level. Annual mean number of attributed cases to chronic bronchitis for cities across Europe in 2000 compared to the projected ones in 2020 per million inhabitants



5.2 First conclusions

In this chapter the potentially beneficial impacts of the reductions of traffic related emissions for the CAFE CLE and MFR scenarios on human health for the target year 2020 were quantified in terms of the expected life expectancy (LE) and chronic bronchitis. Furthermore, due to the fact that the Exposure Response Functions (ERFs) selected for the needs of the current study which relate pollutant concentrations with specific health endpoints such as chronic bronchitis can be treated as linear only long term average concentrations are under consideration.

The comparison of health impacts due to air pollution for the 20 European cities analysed in this study for reference year 2000 and the year 2020 provides valuable information. The most critical issue is the evolution of the potential increase or decrease of the health impacts depending on the adopted scenario. As expected, the 2020 MFR scenario is the most effective regarding air pollution abatement and consequently reduction of YOLL (Years of Life Lost) or cases of chronic bronchitis.

Both for the CLE and the MFR scenarios, the analysis shows larger reductions of the impacts of air pollution on human health due to the local contribution to air pollution at urban hotspots, rather than due to the overall estimated abatement of air pollution at an urban scale. The reason for this is that the predicted reductions of the local scale contribution to air pollution within cities, is larger than the corresponding reduction of the urban background concentrations for the year 2020, for both scenarios. Furthermore, the average (%) abatement efficiency for the CLE and MFR scenarios for the year 2020, which was estimated for the 20 cities is independent of the selected health impact for any pollutant and per million of inhabitants exposed, due to the linearity assumption of the ERFs adopted for the needs of the analysis. As a result of this linearity, the percentage of the estimated YOLL or cases that are avoided over chronic mortality and bronchitis respectively both for the CLE and the MFR scenarios, is in fact a measure of the potential benefit on human health of the scenario.

Tables 5.1 and 5.2 show results regarding the estimated reduction of the YOLL and the cases of chronic bronchitis due to the long term exposure to $PM_{2.5}$ and PM_{10} concentrations for the CLE and MFR scenarios for the year 2020 for the 20 cities.. A simple comparison between the corresponding reductions in the two tables shows that these are virtually the same. The reason for that is that since the ERFs that have been used for the purposes of the current study can be treated as linear, the achieved reductions in the health impacts in terms of two different health endpoints in non-dimensional terms is the same. Regarding the average long term exposure to PM_{10} at an urban scale for the 20 cities considered, for the case of the MFR scenario in the year 2020, there was an approximately 36% reduction of the estimated health impacts, both in terms of YOLL and estimated chronic bronchitis. The corresponding average reduction achieved for the CLE scenario for 2020 was 20.5%. The average abatement efficiency of CLE and MFR scenarios for the 20 cities is higher when it is calculated on the basis of the estimated traffic related concentrations. More specifically, in the case of the MFR scenario for the target year 2020 there was a reduction of 69% in the estimated health impacts. The corresponding reduction for the CLE scenario for the year 2020 was 63.5%. As regards the long term exposure to $PM_{2.5}$ the average achieved reductions in the estimated health impacts for the MFR scenario was approximately 67% while for the CLE scenario it was 37%. As in the case of PM_{10} the average efficiency of CLE and MFR scenarios is higher with regard to traffic contribution street deltas. More specifically, 2020 MFR contributes to an approximately 69.5% reduction of health impacts due to traffic contribution and 2020 CLE contributes to 62.5% reduction.

It should be emphasised that there are inherent uncertainties to the estimates of the health effects due to specific assumptions in the present analysis. General sources of individual uncertainties could be introduced:

- from data series uncertainties,
- emission and atmospheric dispersion and chemistry model uncertainties
- uncertainties regarding the future synergies and idiosyncrasies of the analyst in the interpretation of ambiguous or incomplete information

As regards the uncertainties inside the core of health impact assessments, these usually apply air pollution effect estimates derived from a study in one population (the evidentiary population), to estimate impacts in another (the target population). Such assessments assume that the effect estimates in the evidentiary population are transferable, or that they can be generalised, to the target population. The validity of this assumption implicitly requires that the two populations are similar with regard to factors that influence the magnitude of the effect estimates, such as structure of the morbidity, basic health status, or composition of the air pollution mix (Krzyzanowski *et al*, 2002). That is to say that transferring ERFs to other countries requires data for the respective together with an assumption about the Relative Risk (RR): is the RR the same for populations different from the one in the epidemiological study? On the part of the air pollution mix, separating out the roles of SO_2, NO_2 and PM in the ERFs is particularly problematic, given that they tend to vary together in most locations and studies. It is not clear to what extent the apparent effects of PM are in reality a reflection of effects of NO_2 or SO_2 or vice versa, or whether the presence of other pollutants affects the toxicity of PM. Thus there are uncertainties in applying ERFs in a situation where the ambient pollutant mixture is different from the one where the original epidemiological study was carried out (ExternE, 2005). Moreover, assumptions about causal links between a pollutant and a health impact, slope of an ERF, assumptions about the form of an ERF (e.g. with or without threshold) contributes to the inherent uncertainties in the present analysis.

As a final remark it should be emphasized that in the current state of knowledge, there are still gaps and uncertainties. The purpose of ongoing research is to cover more effects and thus reduce gaps and in addition refine the methodology to reduce uncertainties. Clarity in defining these issues is a prerequisite for proper interpretation of the results in the policy arena. Nevertheless, the results are often prone to misinterpretation, even when the assessment is done carefully, and its multiple uncertainties should be carefully presented and explained to decision makers, the press, and the public (Krzyzanowski et al, 2002).

Table 5.1 Estimated reduction of the LE loss in terms of YOLL due to long term exposure to PM_{10} and $PM_{2.5}$ concentrations for the CLE and MFR scenarios for the year 2020 based on urban background levels

City	Reference Year 2000		2020 CLE		2020 MFR	
	Estimated YOLL		Reduction (%) of YOLL		Reduction (%) of YOLL	
	PM_{10}	$PM_{2.5}$	PM_{10}	$PM_{2.5}$	PM_{10}	$PM_{2.5}$
Antwerp	21556	16727	23	31	41	57
Athens	13808	9867	11	24	30	63
Barcelona	15396	12447	22	42	35	66
Berlin	12132	7807	14	36	25	60
Brussels	18304	15500	24	37	37	61
Budapest	19200	14773	40	58	54	83
Copenhagen	11600	6727	12	30	24	64
Gdansk	12156	6513	20	42	34	75
Graz	11204	6013	10	28	22	60
Helsinki	12124	7540	11	24	28	66
Katowice	23464	17127	34	44	54	74
Lisbon	12224	7927	7	19	26	61
London	14004	9933	15	32	27	58
Marseilles	12800	8053	15	32	32	71
Milan	16912	15300	28	49	41	71
Paris	21772	21540	22	33	45	67
Prague	13900	9433	28	55	38	75
Rome	12860	9227	18	40	32	71
Stuttgart	11260	6787	12	37	22	61
Thessaloniki	12152	7533	9	23	27	65

Table 5.2 Estimated reductions of cases of chronic bronchitis due to long term exposure to PM_{10} and $PM_{2.5}$ concentrations for the CLE and MFR scenarios for the year 2020

City	Reference Year 2000 Estimated cases of chronic bronchitis		2020 CLE Reduction (%) of chronic bronchitis		2020 MFR Reduction (%) of chronic bronchitis	
	PM_{10}	$PM_{2.5}$	PM_{10}	$PM_{2.5}$	PM_{10}	$PM_{2.5}$
Antwerp	1428	1108	23	31	41	57
Athens	915	654	11	24	30	63
Barcelona	1020	825	22	42	35	66
Berlin	804	517	14	36	25	60
Brussels	1213	1027	24	37	37	61
Budapest	1272	979	40	58	54	83
Copenhagen	769	446	12	30	24	64
Gdansk	805	432	20	42	34	75
Graz	742	398	10	28	22	60
Helsinki	803	500	11	24	28	66
Katowice	1554	1135	34	44	54	74
Lisbon	810	525	7	19	26	61
London	928	658	15	32	27	58
Marseilles	848	534	15	32	32	71
Milan	1120	1014	28	49	41	71
Paris	1442	1427	22	33	45	67
Prague	921	625	28	55	38	75
Rome	852	611	18	40	32	71
Stuttgart	746	450	12	37	22	61
Thessaloniki	805	499	9	23	27	65

6 Conclusions and future work

The increasingly tight emission standards generally lead to reductions of both PM and NO_x emissions in 2020, compared to the reference year (2000). The absolute magnitude of the reduction depends on the vehicle category considered. As one would expect, the MFR scenario leads to higher reductions compared to CLE. NO_x emissions seem to decrease in the future for all vehicle categories expect diesel passenger cars (CLE scenario) and power two wheelers (both MFR and CLE scenarios). The increase of NO_x from two wheelers comes as a result of improvements in their combustion efficiency. According to the MFR scenario, power two wheelers will contribute to as much as 60% NO_x of the gasoline passenger car fleet in 2020. The increase in diesel passenger car emissions in the 2020 CLE scenario mostly comes from their increased fleet penetration compared to 2000. In addition, the real-world emission performance of late technology diesel passenger cars seems not to fulfill the emission standards for NO_x emissions. This conclusion is supported by consistent experimental information showing exceedances of NO_x emission standards from Euro 4 diesel passenger cars under real-world driving cycles. This is an important issue that limits the effectiveness of the regulations currently in place.

NO_2 emissions significantly decrease from all vehicle categories in the future, except diesel passenger cars. In this case a significant increase of 3 to 5 times over the 2000 levels is observed for the MFR and the CLE scenarios, respectively. Similar to NO_x, the increase is largely due to the higher penetration of diesel passenger cars into the future. On the other hand, the high NO_2/NO_x ratio observed for late technology diesel cars, equipped with oxidation after treatment devices (oxidation catalysts, catalysed diesel particle filters) may even lead to an increase in absolute NO_2 emission levels in the exhaust, compared to the past. The increase of primary NO_2 emissions from diesel passenger cars may lead to significant implications in meeting air quality targets.

The tight emission standards established for exhaust PM from future diesel passenger cars lead to very effective reductions in the future. When considering both exhaust and non exhaust $PM_{2.5}$ in the MFR scenario, diesel passenger cars become the fourth most important source, following heavy duty vehicles, gasoline passenger cars and power two wheelers. The fulfillment of future PM emission standards with the use of diesel particle filters turns therefore to a success story for the diesel technology.

The modeling methodology applied for the estimation of the air pollution at urban hotspots for both CAFE CLE and MFR scenarios, largely followed the methodology used within the frame of the EEA's Technical Report 1/2006. In particular the complete regional-urban-local scale model cascade application which was used comprised of the OFIS (Arvanitis and Moussiopoulos, 2003) and OSPM (Berkowicz *et al.*, 1997) models. Results from the OFIS model, which was driven by the regional scale EMEP model, served as input for the urban background concentrations necessary for OSPM. The methodology was validated on the basis of comparisons between numerical results and measurements from Airbase within the frame of the 2006 study. Therefore, further validation of the modeling system was beyond the scope of the current report.

In all cases and for all pollutants, compared to the reference year 2000, a significantly reduced street increment is projected according to both the CLE and the MFR scenarios. This reduction is considerably higher for the case of the MFR scenario, for all cities under consideration. The operational model used for the prediction of air pollution levels at urban hotspots, is highly sensitive to both the urban background concentrations and the traffic related emissions at a local scale. It is for that reason that the numerical results for air pollution levels at urban hotspot are overall in line with the projected reductions of both the traffic emissions and the urban scale emissions for all scenarios for the target year 2020. According to the results, both for the CLE and the MFR scenarios, no hourly exceedances of the 2010 limit value of 200 $\mu g/m^3$ (allowed number of exceedances is 18 times per year) for NO_2 are predicted. However for PM_{10}, for the CLE scenario, daily exceedances of the 2010 limit value of 50 $\mu g/m^3$ are predicted in many cities, while in the MFR scenario most cities will have close to zero exceedances with the exception of Antwerp and Paris. As regards the Retrofit scenarios, it is safe to conclude that compared to the MFR scenario, a significant effect was observed only for NO_x and only a small effect for PM. The reason was that as regards PM, the MFR assumes the introduction of aftertreatment technology anyhow while for NO_x this is not the case. Given that non-exhaust emissions can barely be reduced by technical measures any strategy to attain the PM limit values needs ought to be supplemented by traffic planning and economic measures to shift mileage of motor vehicles to other transport modes. This is the more important as vehicle technology is being exhausted by Euro 6/VI, leaving a much higher share to non-exhaust emissions.

Based on the modeling results for the air pollution levels for the CAFE CLE and MFR scenarios for the target year 2020, an analysis of the estimated impacts of the air quality on human health was performed using chronic mortality and chronic bronchitis as the main health endpoints. Results showed that compared to the reference year 2000, for both scenarios for the target year 2020, there was a considerable reduction of the estimated health impacts due to the long term exposure to PM_{10} and $PM_{2.5}$. The average abatement efficiency for the two scenarios averaged over the 20 cities under cosideration ranged:

- from a minimum of 20.5% for CLE to a maximum of 36% for MFR based on the long term exposure of PM_{10}
- from a minimum of 37% for the CLE scenario to a maximum of 67% for the MFR scenario based on long term exposure to $PM_{2.5}$

The issue of the negative impacts of the urban air quality on human health has been receiving increased attention. In fact evidence of the adverse health effects of the degraded air quality in densely populated urban areas is alarming. It is therefore necessary to develop and apply in the future a more accurate methodology for air quality assessment, capable of taking into consideration the effects of the intra-urban differences. At the same time, new more realistic scenarios need to be developed and analyzed which take into account the effects of climate change on the evolution of traffic emissions. In the same sense and for considering the climate change effects, it will be helpful to develop updated long term forecasting tools for the meteorological conditions in the future at all scales

7 References

Abbey, D E, Lebowitz, M D, Mills, P K, Petersen, F F, Beeson, W L, and Burchette, R J 1995b. "Long-term ambient concentrations of particulates and oxidants and development of chronic disease in a cohort of nonsmoking California residents". Inhalation Toxicology, vol. 7,19-34.

AEAT 2007. "The impact of changes in vehicle fleet composition and exhaust treatment technology on the attainment of the ambient air quality limit value for nitrogen dioxide in 2010". DG Environment study, currently in draft-final stage. Data submitted to the project by Melanie Hobson.

AIRBASE 2007. The European Airquality database. http://air-climate.eionet.eu/databases/airbase/

AIRNET 2007. A Thematic Network on Air Pollution and Health. Internet: http://airnet.iras.uu.nl.

Amann, M, Cabala, R, Cofala, J, Heyes, Ch, Tarrason, L, Simpson, D, Wind, P and Jonson, J E 2004. The "Current Legislation" and the "Maximum Technically Feasible Reduction" cases for the CAFE baseline emission projections, Background paper for the meeting of the CAFE Working Group on Target Setting and Policy Advice.

ARTEMIS 2007. Assessment and Reliability of Transport Emission Models and Inventory Systems. Project funded by the European Commission (DG Transport) in the 5th Framework Programme. More information on http://www.trl.co.uk/artemis/.

Arvanitis, A and Moussiopoulos, N 2003. Coupling an aerosol module to a deterministic long term ozone exposure model. 26th NATO/CCMS ITM, Istanbul, Turkey.

Berkowicz, R, Hertel, O, Larsen, S E, Sørensen, N N, Nielsen, M 1997. Modelling traffic pollution in streets. Ministry of Environment and Energy, National Environmental Research Institute, Denmark.

CAFE 2007. Clean Air for Europe program website. Internet: http://ec.europa.eu/environment/archives/cafe/general/keydocs.htm.

Cofala J 2004. private communication.

COPERT, 2006. Computer Programme to Calculate Emissions from Road Transport. Available online at http://lat.eng.auth.gr/copert.

Cuvelier, C, Thunis, P, Stern, R, Moussiopoulos, N, Builtjes, P, Rouil, L, Bedogni, M, Tarrason, L, Amann, M and Hayes, C 2004. City-Delta Phase 2, Draft Final Report December version.

Dockery, D W, Pope C A, Xu, X, Spengler, J D, Ware, J H, Fay, M E, Ferris, B G and Speizer, F E 1993. "An association between air pollution and mortality in six US cities". New England J of Medicine, vol.329, p.1753-1759 (Dec. 1993).

EEA 2006. Air pollution at street level in European cities, EEA Technical Report No 1/2006. http://air-climate.eionet.europa.eu/reports/EEA_Techn_Rep_1_2006

EMEP 2007. European Monitoring and Evaluation Programme. http://www.emep.int.

ExternE 2005. Externalities of Energy, "Methodology 2005 Update", Sustainable Energy Systems, Directorate-General for Research, ISBN 92-79-00423-9 European Communities.

Friedrich, R, Reis, S (eds.) 2004. Emissions of Air Pollutants — Measurements, Calculation, Uncertainties — Results from the EUROTRAC Subproject GENEMIS. Springer Publishers, ISBN 3540008403.

Gense, N L J, Riemersma, I, Such, C, Ntziachristos, L, 2006. Euro VI technologies and costs for Heavy Duty Vehicles. The expert panels summary of stakeholders responses. TNO report 06.OR.PT.023.2/NG, Delft, Netherlands, p. 56.

Hausberger, S 2006. Emission Levels of Diesel Cars EURO 1 – EURO 4. Results from the EU 5FP ARTEMIS project. Results available at the European Federation for Transport & Environment website. http://www.transportenvironment.org/Article248.html.

Hoek, G, Brunekreef, B, Goldbohm, S, Fischer, P and van den Brandt, P A 2002. "Association between mortality and indicators of traffic-related air pollution in the Netherlands: a cohort study". The Lancet, Vol.360 (Issue 9341, 19 October), 1203-1209. Seealso "Mortality and indicators of traffic-related air pollution", Lancet, Vol.361, Issue 9355, 1 February 2003, Page 430.

Holland, M, Hunt, A, Hurley, F, Navrud, S, Watkiss, P, 2005a. Methodology for the Cost-Benefit Analysis for CAFE: Volume 1: Overview of Methodology. Didcot. UK: AEA Technology Environment.

Hurley, F, Hunt, A, Cowie, H, Holland, M, Miller, B, Pye, S, Watkiss, P 2005a. Methodology for the Cost-Benefit Analysis for CAFE: Volume 2: Health Impact Assessment. Didcot. UK: AEA Technology Environment. Available: http://europa.eu.int/comm/environment/air/cafe/pdf/cba_methodology_vol2.pdf.

IIASA 2007. Rains model website, by the IIASA at http://www.iiasa.ac.at/web-apps/tap/RainsWeb/. Registration required for access.

Krewski, D, Burnett, R T, Goldberg, M S, Hoover, K, Siemiatycki, J, Abrahamowicz, M, White, W H et al. 2000. "Particle Epidemiology Reanalysis Project". Health Effects Institute, Cambridge MA. Available at http://www.healtheffects.org/Pubs/particle.htm.

Krzyzanowski, M, Cohen, A, Anderson, R 2002. WHO Working Group, "Quantification of health effects of exposure to air pollution." Occupational and environmental medicine, 59 (12), p.791-793.

Künzli, N, Kaiser, R, Medina, S, Studnicka, M, Chanel, O, Herry, M, Horak, F, Puybonnieux-Texier, V, Qunel, P, Schneider, J, Seetaler, R, Vergnaud J C and Sommer H 2000. "Public health impact of outdoor and traffic-related air pollution: a European assessment", Lancet, 356 (Sept.), 795-801.

Mayer, A, Kasper, M, Mosimann, Th, Legerer, F, Czerwinski, J, Emmenegger, L, Mohn, J, Ulrich, A, Kirchen, P 2007. Nanoparticle-emission of Euro 4 and Euro 5 HDV compared to Euro 3 with and without DPF. SAE Technology paper 2007-01-1112.

MERLIN 2007. Multi-Pollutant Multi-Effect Modeling of European AiR Pollution ControL Strategies - an INtegrated Approach. http://www.merlin-project.de/main.html

Moussiopoulos, N, Kalognomou, E A, Mellios, G, Samaras, Z, Larssen, S, Gjerstad, K I, van den Hout, D, Teeuwisse, S, de Leeuw, F, Taylor, M and Papathanasiou, A 2004. ETC/ACC Street Emission Ceilings (SEC) exercise Phase 1, ETC/ACC Technical Paper 2003/11, August 2004.

Moussiopoulos, N, Kalognomou, E A, Larssen, S, Gjerstad, K I, Mellios, G, Samaras, Z, van den Hout, D, Teeuwisse, S, van Aalst, R, Vouitsis, I, Papathanasiou, A, Eleftheriadou, S, Barmpas, Ph and Vlachokostas, Ch 2005. ETC/ACC Street Emission Ceilings (SEC) exercise Phase 2, ETC/ACC Technical Paper 2004/5.

Ntziachristos, L, Boulter, P 2003. Road vehicle tyre and break wear and road surface wear. Chapter B770 of the 2006 EMEP/CORINAIR Emission Inventory Guidebook. Technical report No 11/2006, European Environment Agency, Copenhagen, Denmark.

Pope, C A III, Thun, M J, Namboodiri, M M, Dockery, D W, Evans, J S, Speizer, F E and Heath, C W Jr 1995. Particulate air pollution as predictor of mortality in a prospective study of US adults. Am J Resp Crit Care Med 151: 669-674.

Pope, C A, Burnett, R T, Thun, M J, Calle, E E, Krewski, D, Ito, K and Thurston, G D 2002. “Lung cancer, cardiopulmonary mortality, and long-term exposure to fine particulate air pollution” J. Amer. Med. Assoc., vol.287(9), 1132-1141.

Rabl, A 2003. “Interpretation of Air Pollution Mortality: Number of Deaths or Years of Life Lost?” Journal of Air and Waste Management, Vol.53(1), 41-50 (2003).

Schwarz, U 2002. Methoden und Modelle zur multiskalig hoch aufgelösten Bestimmung von luftverunreinigenden Emissionen in Europa. Dissertation am Institut für Energiewirtschaft und Rationelle Energieanwendung, Universität Stuttgart. (Doctoral thesis at the Intitute of Energy Economics and the Rational Use of Energy, University of Stuttgart) (in German).

Smit, R 2007. Primary NO_2 emission factors for local air quality assessment in the Netherlands. Personal communication.

TREMOVE 2007. policy assessment model to study the effects of different transport and environment policies on the transport sector for all European countries. More information available online at http://www.tremove.org/.

Wiartalla, A, Körfer, Th, Schnitzler, J, Nisius, P 2007. Diesel NO_x Aftertreatment Concepts with Special Regard to Fuel Consumption and DPF Integration. MinNO_x conference, 01. - 02. February 2007, Berlin, Germany.

Wickert, B 2001. Berechnung anthropogener Emissionen in Deutschland für Ozonvorläufersubstanzen — Modellentwicklung und Sensitivitätsstudien. Dissertation am Institut für Energiewirtschaft und Rationelle Energieanwendung, Universität Stuttgart (Doctoral thesis at the Intitute of Energy Economics and the Rational Use of Energy, University of Stuttgart). (in German).

Wilson, R, Spengler, J 1996. Particles in our air: concentrations and health effects. Boston: Harvard University Press.

WHO 2003. Health aspects of air pollution with particulate matter, ozone and nitrogen dioxide,Report on a WHO Working Group, Bonn, Germany, 13-15 January 2003. World Health Organisation: http://www.euro.who.int/document/e79097.pdf, accessed November 2004.

8 Glossary

AEIG:	Air Emissions Inventory Guidebook
Aspect ratio (W/H):	Width of Street / Average Height of Buildings
Associations:	The statistical correlations between air pollution and a health impact
CAFE:	Clean Air For Europe
Chronic bronchitis:	Reporting chronic cough or sputum on most days, for a period of at least three months of the year and for at least two years
CLE:	Current LEgislation. The CLE scenario corresponds to the future road transport emission evolution, assuming that no further regulatory steps will be brought, further to the emission standard regulations already in place today.
COPERT:	COmputer Programme to calculate Emissions from Road Transport
CRT:	Continuously Regenerating Trap
DPC:	Diesel Passenger Cars
DPF:	Diesel Particle Filter
EMEP:	A policy driven program under the Convention on Long-range Transboundary Air Pollution for international co-operation to solve transboundary air pollution problems.
Emission factor:	A value which is multiplied with activity data (e.g. vehicle-km or tones of fuel consumed) to calculate total emissions. For road transport this is specific to fuel, combustion and aftertreatment technology.
ERF:	Exposure Response Function
Euro 1 – 6:	Level of emission control technology for passenger cars
Euro I – VI:	Level of emission control technology for heavy duty vehicles
FP5:	Fifth Framework Programme
GPC:	Gasoline Passenger Cars
GVW:	Gross Vehicle Weight
HDV:	Heavy Duty Vehicles
LDV:	Light Duty Vehicle
LE:	Life expectancy
Local scale:	Spatial scale which is typically used to resolve flow and dispersion over a street or a cluster of streets
LPG PC:	Liquefied Petroleum Gas Passenger Cars
MCYCLES:	Motorcycles
MFR:	Maximum Feasible Reduction. The MFR scenario corresponds to what is expected to be achieved with all legislation that is currently in place and with the foreseeable regulatory steps for the near future.
OFIS:	Ozone Fine Structure
OSPM:	Operational Street Pollution Model
SCR:	Selective Catalytic Reduction
Regional scale:	Scale which typically resolves meteorological and transport phenomena which cover a medium sized country or a cluster of small to medium sized countries
Relative Risk (RR):	The ratio of the incidence observed at two different exposure levels
Street increments:	Difference between street and urban background concentrations
TREMOVE:	A policy assessment model to study the effects of different transport and environment policies on the transport sector for all European countries
YOLL:	Years of Life Lost
Urban scale:	Spatial scale associated with meteorological and transport phenomena which extend to a size of city and its surrounding areas
Urban hotspots:	Streets or locations in cities which suffer from elevated levels of air pollution due to increased traffic emissions and/or inefficient ventilation

Urban background: Ambient pollutant concentrations over a city