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Analysis of model responses to emission-reduction scenarios within the CityDelta project

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Abstract

This paper investigates how air quality models applied at different scales (50 and 5 km horizontal resolutions) can predict pollution levels in response to emission control strategies in various cities in Europe. This study, involving five modelling teams and focused on four European cities, has been conducted within the CityDelta project (http://aqm.jrc.it/citydelta). The CityDelta models generally agree, on the O_3 changes expected from scenarios representative of the current legislation on air pollution in 2010, named CLE. They also agree about less scope for further improvements from emission controls beyond CLE. For PM_{10}, more significant differences between the models are observed, especially between models with different spatial resolutions. However, these differences are city-dependent and are larger in complex geographical areas such as Milan in the Pô Valley than in the Paris area.

Fine scale models generally capture important urban scale effects, which are not represented by regional scale models. For instance, they improve the simulation of potential O_3 increase caused by NO_x emissions reduction in NMVOC-limited regime situations. Large scale models generally underpredict PM mean concentrations in city areas. A series of emission scenarios to address the question of the efficiency of local emission controls designed independently from regional measures is analyzed. The analysis of the CityDelta results contributes to the quantification of the impact of grid resolution in air quality modelling, and its application to emission control scenarios.

Keywords: Air quality; Ozone; PM; Model intercomparison; Emission-reduction scenarios

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1. Introduction

Air pollution has been one of Europe's main political concerns since the late 1970s. The clean air for Europe programme (CAFE) which has been launched in March 2001 is a programme of technical analysis and policy development that underpinned the development of the Thematic Strategy on Air Pollution under the Sixth Environmental Action Programme, which has been adopted by the European Commission in September 2005. It provides the basis for an integrated policy to protect human health and environment against significant negative effects of air pollution, taking into account economic constraints. It mainly focuses on ozone (O_3) and particulate matter (PM).

The World Health Organization (WHO) has recently established new evidence regarding the role of O_3 in the increase of premature deaths (WHO, 2003). Both peak values and long-term exposure have been recognized to have a negative effect on human health. Recent research has also shown that an increase of $10\,\mu g\,m^{-3}$ (compared to background values ranging from 20 to $45\,\mu g\,m^{-3}$ in urban areas) in the daily maximum 8-h mean leads to an increase of 0.3% of the relative risk for all-cause mortality (Ross Anderson et al., 2004). Thresholds above which health impact becomes significant are defined and regularly revised by the WHO.

PM is emitted from anthropogenic combustion processes and natural sources but also formed from physico-chemical mechanisms involving gas phase and solid components. Fine particulates ($PM_{2.5}$) are considered to be responsible for increased mortality over Europe. Anthropogenic $PM_{2.5}$ levels are expected to be responsible for a loss of ten months of life expectancy in Benelux and Po-Valley by 2020, in spite of application of the current legislation devoted to air pollution control (Amann et al., 2005a). It is also recognized that adverse effects from PM long-term exposure occur whatever the concentration levels are (WHO, 2006; Pope et al., 2002).

The CityDelta project is an open modelling exercise designed to understand the changes in urban O_3 and PM concentrations in response to emission-control scenarios. It aims at analyzing the differences between responses delivered by a set of models and at characterizing the variability of their results. The variability of model responses is actually a compound variability of the scientific uncertainties reflected by different model constructions and the imprecision engendered by the incomplete harmonization of model inputs. Its evaluation may be seen as a first step of an uncertainty analysis of modelling tools.

The focus of CAFE, clearly oriented towards human exposure, led to consider the city scale as the domain of main interest. With respect to this purpose, the CityDelta project focused on several European cities, chosen to be representative of distinct air pollution patterns. Eight cities were studied in a first stage by 16 modelling teams. In a second stage defined to deepen the analysis of PM results, six modelling teams delivered results for four of the initial eight cities: Berlin, Milan, Paris, and Prague. The main results of CityDelta are presented in an overview paper (Cuvelier et al., 2006), whereas the evaluation of the model results against observations is discussed in more detail in Vautard et al. (2006). The present paper addresses the sensitivity studies to assess the impact of emission control scenarios. After a brief description of the methodology (Section 2), the following topics will be discussed: first, a comparison between models run with a coarse grid resolution (50 km) and models run with a fine grid resolution (lower than 5 km) is performed for the 2000 reference base case over the different cities (Section 3). In the sections below, the former type of runs will be referred as "large scale" (LS) and the latter as "fine scale" (FS). In Section 4 the variability of FS and LS responses to emission scenarios in 2010 and beyond is investigated. In Section 5 the model responses to local scale emission reductions are compared with responses to regional emission reductions. Finally, conclusions are drawn in Section 6.

2. Methodology

The analysis of model responses to different emission scenarios foreseen in 2010 and beyond is carried out over four different cities: Berlin, Milan, Paris and Prague. These cities were selected considering their large size, their representativeness of various European climates, and the availability of emission inventories and observations. Only results from models participating to the so-called second phase of the CityDelta project are considered in this paper.

A simulation domain of 300 km × 300 km around each city was imposed on the modelling groups. Model results have been classified into two different

groups according to their spatial resolution: LS and FS models with a horizontal grid spacing equal or smaller than 50 and 5 km, respectively. The following six models participated: REM-CALGRID (Stern et al., 2003), CHIMERE (Schmidt et al., 2001; Bessagnet et al., 2004), and LOTOS (Schaap et al., 2004 and references therein) all covering both the LS and FS resolutions, OFIS (Moussiopoulos and Douros, 2005) and CAMx (ENVIRON, 2004) operating at the FS, and EMEP (Simpson et al., 2003) limited to the LS. Only Milan was covered by the CAMx simulations. A detailed description of the model features is given in Cuvelier et al. (2006) and Vautard et al. (2006). Each model used its own meteorological pre-processor and boundary conditions. Some obtained these (REM, CHIMERE and LOTOS) from regional-scale simulations, while others (CAMX and OFIS) used values calculated with the Unified EMEP Eulerian model.

All models used a similar anthropogenic emission inventory, interpolated and adapted to their grid configuration and chemical mechanism. Year 2000 has been chosen as base case year because of availability of both local and regional scales emission inventories. For model simulations at regional scale, emissions were based on the EMEP/TNO inventories (Vestreng, 2003). For city-scale emissions, high-resolution inventories were provided by local organization in charge of air quality monitoring. For some cities, this inventory did not cover the whole 300 km × 300 km required simulation area, in which case EMEP emissions were used as a complement. For some species and some cities, city-scale and regional-scale emissions were not consistent, especially for non-methane volatile organic compounds (NMVOC) and PM, due to differences in the top-down and bottom-up building methodologies. A consistent set of emissions at FS and LS has been built by scaling the local inventories to their EMEP counterparts in the following manner: for each city, emitted species and activity sector the city total (calculated on the area covered by the high-resolution emission inventory) has been scaled to its EMEP counterpart. For Paris and Berlin where the fit between regional and local scale emissions was best, this scaling has even been done on a (EMEP) grid-by-grid basis, leading to a slight spatial re-allocation of the emissions. Due to the grouping of emissions in different activity sectors at the city and regional scales correspondences across sectors have been defined for each city (with the exception of Milan where activity sectors were defined consistently across scales). For species not provided at the city resolution, EMEP values have been used. Finally, because biogenic emissions strongly depend on various factors, e.g. temperature and land use, each model generated its own biogenic emissions.

Although emissions are based on their 2000 estimate, meteorology has been computed for 1999 which was considered as a more representative year from a climatological point of view. Meteorological pre-processors were run for the whole year 1999 on an hourly basis, with spatial resolutions compatible with those of the chemical models.

Emission scenarios for 2010 have been prepared by the CIAM (Center for Integrated Assessment Modelling) with emission reductions applied depending on the country, the pollutant and the activity sector. In CityDelta, activity sectors included traffic and stationary sources differentiated in high (above 50 m) and low level sources. The scenarios were constructed from two sets of emission projections related to year 2010 (Amann et al., 2005b):

- the current legislation emission (CLE) which assumes emission reductions in all activity sectors related to the present legislation adopted in the European Union (25 countries);
- the maximum feasible reduction (MFR) which assumes full implementation of the most advanced technical emission control measures which should be available in 2010, although excluding premature retirement of existing equipment before the end of its technical life time . These measures concern point sources as well as mobile sources.

The 2010 CLE and 2010 MFR emission scenarios act simultaneously on sulfur oxides (SO_2), nitrogen oxides (NO_x), NMVOC, primary particulate matter (PPM), carbon monoxide (CO), and ammonia (NH_3). Two supplementary MFR-like scenarios consider reductions limited to specific group of pollutants. The so-called MFR1 scenario addresses combined reductions of NO_x, SO_2 and PPM, whereas the so-called MFR2 scenario deals with NMVOC, NH_3 and CO reductions. For each of these scenarios, emission reductions are applied consistently at the regional and city scales. Moreover, in order to assess the effectiveness of emission control strategies imposed at the city level (i.e. independently of the regional measures) three

additional emission control situations have been designed. They impose MFR, MFR1 and MFR2 reductions restricted to the 300×300 modelling domain while keeping boundary conditions at CLE levels. Biogenic emissions remain unchanged in all 2010 scenario calculations.

Comparison of model responses across cities is hampered by the fact that both the CLE and MFR emission controls differ from a country to another. This is illustrated in Table 1 which indicates for each city, relative emission changes for NO_x, NMVOC and PPM_{10} from the current 2000 situation to CLE and from CLE to MFR.

In most of this work, syntheses of the results are presented and interpreted using two types of results: a models ensemble response corresponding to the average of all model results, and an associated model variability defined as the difference between the maximum and minimum model responses. Delle Monache and Stull (2003) and McKeen et al. (2005) show in their paper that this average generally gives more satisfactory results in comparison with observations than models taken individually. It is assumed that this model variability provides a representation of the uncertainty inherent to air quality models construction but also encompasses the imprecision due to the incomplete harmonization of model inputs: boundary conditions, biogenic emissions, meteorological fields.

Conclusions are obviously driven by the type of indicator considered for interpretation. In some cases, a non-zero threshold within the indicator can increase the model sensitivity and consequently the variability among model results. For example, the so-called AOT40 and AOT60 (accumulated over threshold 40 or 60 ppb) which represent the hourly O_3 concentration exceeding 40 (resp. 60) ppb accumulated over summer time, are very sensitive. Indeed model responses may differ only by a few ppb around the threshold value and exhibit large differences when AOTs are considered. In this work, two different types of indicators are used: the summer (April–September) or annual averages, and the SOMO35 (Sum Of maximum daily 8-h Means Over 35 ppb) (Amann et al., 2005a, b).

Table 1
Percentage NO_x, NMVOC and PPM_{10} emission reductions considered for the 2000 to CLE (bold) and CLE to MFR (within brackets) emission scenarios for each city

	NO_x	NMVOC	PPM_{10}	$PPM_{2.5}$
Berlin	**33** (8)	**36** (7)	**55** (11)	**47** (33)
Milan	**28** (19)	**38** (21)	**40** (26)	**44** (27)
Paris	**26** (19)	**37** (19)	**48** (23)	**47** (26)
Prague	**34** (33)	**3** (42)	**54** (28)	**47** (33)

Interpretation of the results has also been conducted with special care to the spatial area over which model results are averaged and compared. Two domains covering different areas are considered in this study. The "city domain" (CD) is restricted to an area covering $50 \times 50\,km^2$ centered on the city of interest, while the "city center" (CC) area covers a FS grid cell of $5 \times 5\,km^2$ in the core of the urbanized area.

3. Large and fine scale modelling for the base case year

One main purpose of CityDelta is to understand which additional information may potentially be gained by applying FS models at the urban scale. The purpose of this section is to analyze how differences between LS and FS models vary across cities and how robust are both the FS and LS model predictions (analysis of the model variability). To address these points FS and LS model ensembles are constructed and their behavior compared over the four cities (Fig. 1). The analysis deals with the differences among models themselves, and leaves aside a possible evaluation of the model results against observations. This topic is the subject of a companion publication (Vautard et al., 2006).

The model responses are analyzed over the CD (i.e. $50 \times 50\,km^2$). The results given by the LS and FS model ensembles for mean 24 h-average summer O_3 (Fig. 1a) and yearly average PM (Fig. 1c) are quite similar. However, LS modelling generally leads to: (1) an overestimation of O_3 concentrations presumably due to the larger dilution of NO_x emissions. This dilution limits titration effects at urban sites and reduces the production of inert NO_z compounds, process which generally limits O_3 production because of OH chain shortening and (2) an underestimation of the PM levels.

In all cases, both the difference between LS and FS model ensembles and the associated model variability are the largest in Milan. This is most probably due to the variability between meteorological inputs, pointing out the difficulty of simulating low wind speeds and frequent thermal inversions characterizing this complex geographical area (Dosio et al., 2002; Minguzzi et al., 2005).

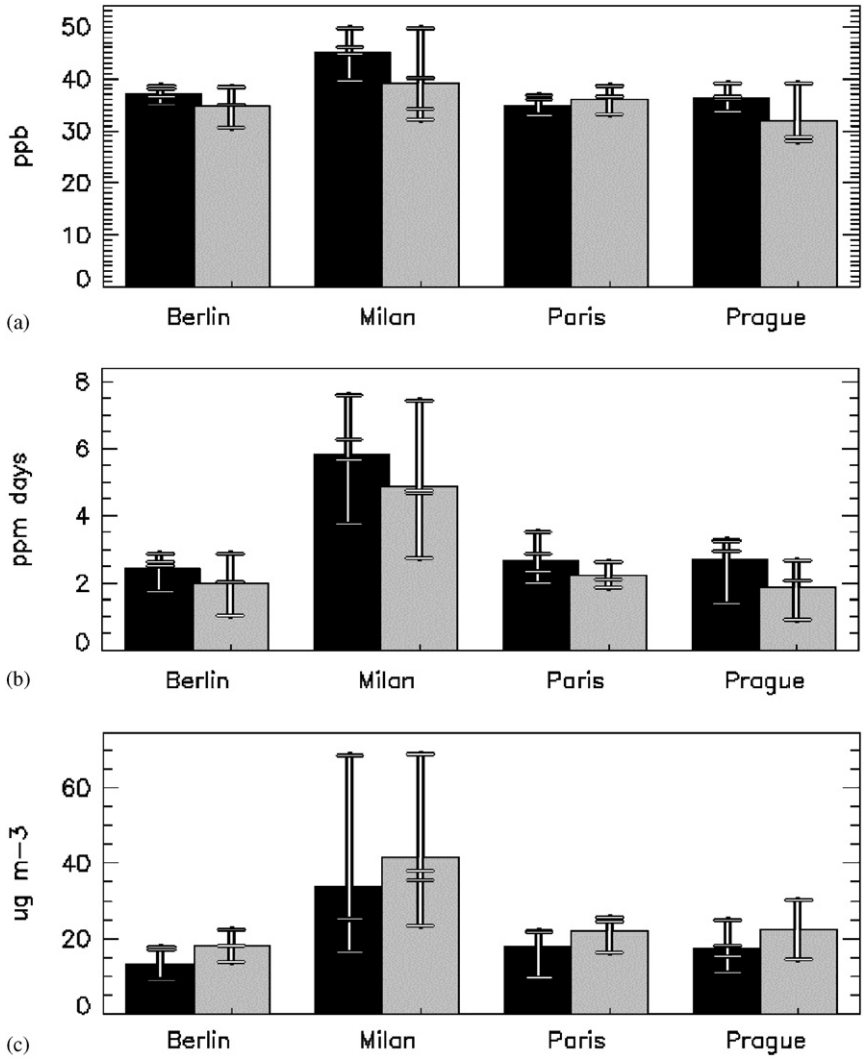

Fig. 1. Comparison of (a) mean 24 h-average summer O_3 (b) SOMO35 and (c) PM_{10} annual mean averaged over the city domain ($50 \times 50\,km^2$) area for the LS (black) and FS (gray) model ensembles. Vertical lines are drawn between the maximum and minimum model responses for each city and model scale to provide an indication of the model results variability. Individual model results are indicated with small intersecting horizontal lines.

Differences between FS and LS model ensemble values are generally smaller than the variability associated to the FS model predictions. It is about 2–3 ppb for O_3 and 5–10 $\mu g\,m^{-3}$ for PM. For all cities the variability characterizing PM results is much larger than the variability characterizing the O_3 results (especially in Milan). This can be explained by the lack of maturity of PM modelling, resulting in a large variety of physical and chemical parametrizations used in the different models. As expected, the model variability increases with accumulated indicators defined with thresholds values as clearly illustrated in Fig. 1b for SOMO35.

Conclusions may be driven not only by the choice of the indicator but also by the choice of the spatial area over which results are averaged for comparison. In the cases studied, when the spatial area over which model results are aggregated gets larger, LS and FS models tend to behave more similarly. Similarly to Figs. 1a and c, Figs. 2a and b show the O_3 and PM_{10} values, respectively, obtained with the LS and FS model ensembles averaged over the CC ($5 \times 5\,km^2$). With the exception of Paris, LS–FS differences between the results averaged over the CD are reduced by two-thirds in all cities as compared to the CC averaging area. The lower

spatial variability of the emissions for the CD area partly explains this fact. Paris is less concerned because city emissions are quite evenly shared over the CD. The same remark holds for PM (Fig. 2b) which shows a clear FS added value in Milan, and Paris and a moderate signal in Berlin and Prague. Another possible explanation for these inter-city urban signal differences lies in the relative contribution of the regional background to the urban levels. In cities where the regional contribution dominates, differences between LS and FS models are expected to be less significant. This point will further be addressed in Section 5.

Finally, Fig. 3 illustrates the FS–LS model differences in terms of the different PM constituents averaged over the CD. The largest differences observed for PM_{10} in the Milan area originate predominantly from the ammonium component, whereas for the Paris and Prague regions, PM_{10} differences arise mainly from the nitrate and sulfate components, respectively, pointing out to the specificity of the local emission inventories. This also reflects the specificity of each region and the difficulty of drawing general conclusions from the set of four cities investigated in the frame of this study.

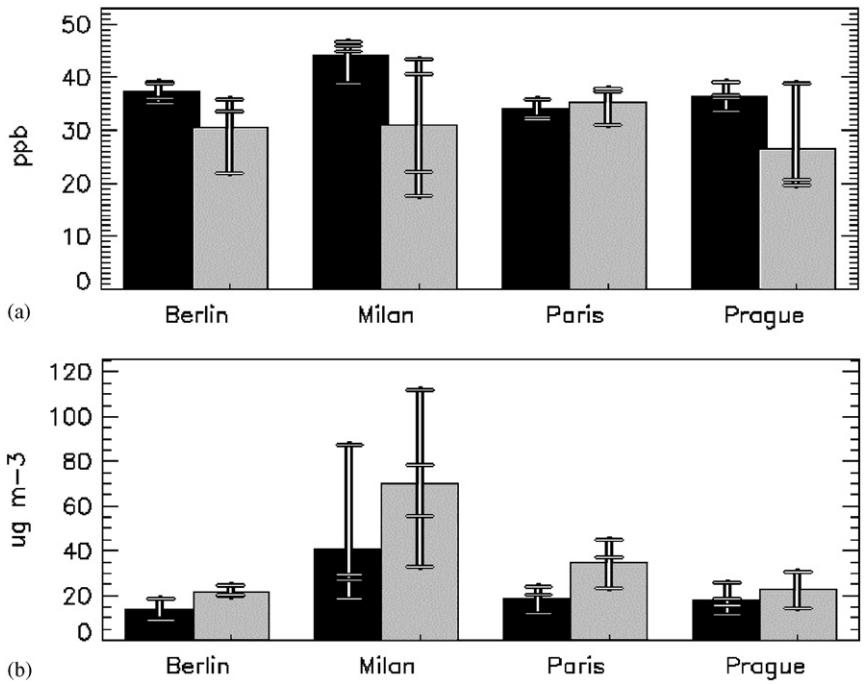

Fig. 2. Same as Fig. 1 for (a) mean 24 h-average summer O_3 and (b) PM_{10} annual mean but model ensemble are here averaged over the city center ($5 \times 5\,km^2$).

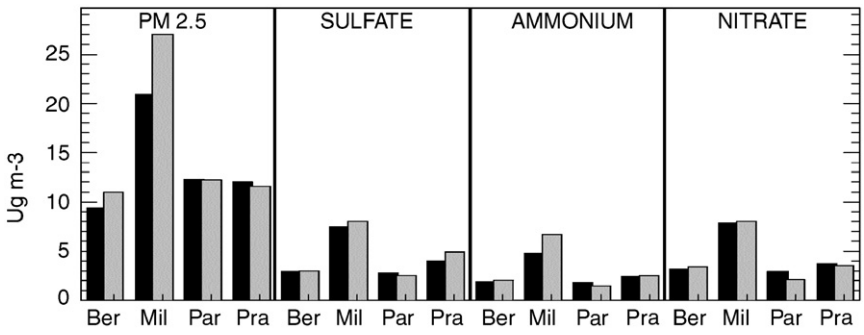

Fig. 3. Comparison of yearly $PM_{2.5}$, sulfate, ammonium and nitrate components for the LS (black) and FS (gray) model ensembles averaged over the city domain (Ber, Mil, Pra, Par stand for Berlin, Milan, Prague and Paris, respectively).

4. Variability of fine and large scale model responses to emission scenarios

The next step of the delta analysis deals with the interpretation of the O_3 and PM model concentrations obtained in the CD in response to NO_x, NMVOC and PPM emission controls. This analysis refers implicitly to the differences between the physico-chemical parametrizations implemented in the participating models. In addition, the efficiency of the emission control scenarios described in Section 2 considered will be discussed.

PPM emission reductions are considered simultaneously with the NO_x reductions in the MFR1 scenario. Results given by the various FS and LS models are presented for the O_3 summer mean, SOMO35 and PM_{10} annual mean (Fig. 4) in terms of the following emission reductions: 2000 to CLE, CLE to MFR1 and CLE to MFR2 for each city (named CLE, MFR1 and MFR2 in Fig. 4).

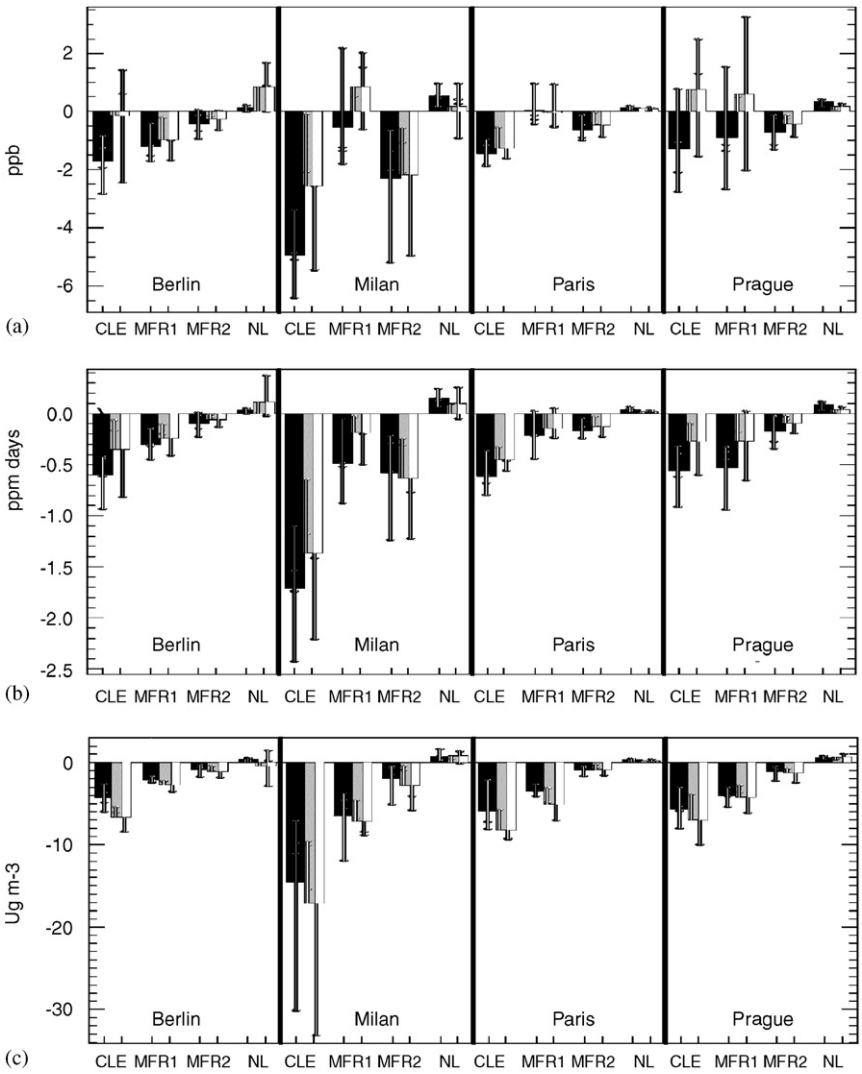

Fig. 4. Comparison of emission reduction impacts for (a) mean 24 h-average summer O_3, (b) SOMO35 and (c) PM_{10} annual mean averaged over the city domain ($50 \times 50\,km^2$) for the LS (black) and FS (gray) model ensembles. CLE, MFR1, MFR2 refer to model responses resulting from 2000 to CLE, CLE to MFR1 and CLE to MFR2 emission reductions, respectively. The NL (non-linear) emission reduction impact for NO_x/NMVOC reductions is the difference between the impact of NO_x–NMVOC reductions from CLE to MFR levels applied simultaneously and the cumulated impact of the same NO_x–NMVOC reductions applied independently from each other. Other features are similar to Fig. 1.

In addition, an estimate of the non-linearity of the NO_x–NMVOC reductions is provided (Fig. 4, column NL) by taking the difference between the impact of combined (MFR to CLE) NO_x and NMVOC emission reductions and the cumulated impact of the two reductions applied independently from each other. The most significant impacts are obtained by reducing the current emissions to their 2010 reference level, named CLE. Emission reductions beyond 2010 have a less noteworthy effect. This is a general trend confirmed by all models for all indicators, even though the range of responses is sometimes rather large. With the exception of Berlin where NO_x controls are more efficient than NMVOC controls in reducing O_3 mean levels (Fig. 4a), other cities show a net benefit from NMVOC controls, whereas NO_x controls lead to disbenefits for most FS models. On the other hand, NO_x controls are more effective than NMVOC ones to bring PM and SOMO35 (Figs. 4c and b) levels down, indicating the positive impact of these controls on the high O_3 concentrations range.

For O_3, a large model variability is obtained for NO_x and NMVOC scenarios. It should be noted that in some cases the variability among the model responses is high, sometimes larger than the magnitude of the average reduction itself. Generally, a wider range of model responses is obtained for NO_x than for NMVOC emission reductions (with the exception of Milan). In particular, the expected urban O_3 increase associated to NO titration and OH chain shortening is not simulated by all models. Beyond differences in chemical parametrizations, the heterogeneity in boundary conditions and meteorological inputs also generates part of the model responses variability. Strong non-linearities in the chemical mechanisms are accounted for by the models (Fig. 4, column NL).

LS models generally overpredict the impact of the CLE emission reductions on O_3 levels by a couple of ppb. These values are consistent with those of Arunachalam et al. (2006) who found a variability of about 1–3 ppb using comparable grid resolutions (4 and 36 km). The same conclusion holds for the MFR1 emission reduction. Differences between scales are less pronounced in the case of NMVOC reductions (MFR2) than for NO_x controls (where titration plays an important role), but are however significant in Berlin or Prague.

It is interesting to note that the variability of model results for PM is quite comparable across scales. The important contribution of the primary components in the PM_{10} concentrations which supposes a larger effect of transport processes compared to the chemical ones, might explain this result. Reduction in model concentrations resulting from the MFR reductions are of the same order of magnitude as those from the CLE reductions, implying that this scenario should be more effective for PM than for O_3. Once more Milan remains different from the other cities, with very large differences among the model results. As commented in the previous section, the complexity of the meteorological situation characterizing the city might be invoked.

5. Regional versus local emission controls

The elaboration of efficient emission control strategies requires a correct knowledge of the respective contributions of local and regional sources to urban pollution levels. The regional background levels of O_3 and PM play a significant role in this respect, since they determine the part of pollution that cannot be regulated at the local scale. In CityDelta a series of emission-reduction scenarios concentrated on the city modelling domain ($300 \times 300\,km^2$) has been designed to investigate the effectiveness of local versus regional control measures. Thus, for each city, three different situations are identified according to the spatial area where emission reductions take place:

- *Urban*: With MFR-like scenario applied over the city modelling domain while keeping CLE conditions outside.
- *Total*: With MFR-like scenario applied all over Europe.
- *Background*: Defined as the difference between total and urban. Assuming linearity, this may be interpreted as the impact of emissions reduced everywhere except over the city modelling domain.

For mean summer O_3 (Figs. 5 and 6) these three components are compared for each city for the FS model ensemble along a 300 km (60 grid cells) East–West transect crossing the modelling domain through the CC for the MFR1 and MFR2 scenarios, respectively. As mentioned in ''Methodology'', MFR emission reductions are not quantitatively similar for all cities, fact that complicates the interpretation of the model responses. To facilitate the interpretation and comparability of

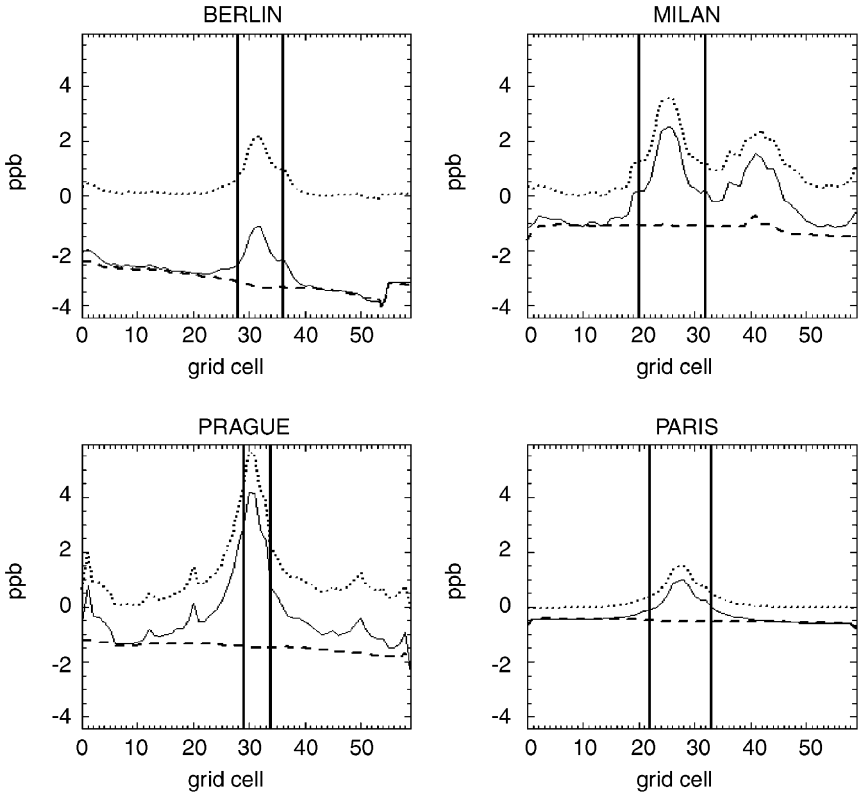

Fig. 5. Modelled impacts of MFR1 (NO_x, SO_2 and PPM) emission reductions on summer mean O_3 concentrations along an East–West transect crossing the city center (delimited by the vertical lines). The urban component (dotted line) results from MFR1 type emission reductions applied over the modelling domain only, whereas the total component (solid line) is obtained when these emission reductions are applied consistently across the whole domain (Europe). The background component (dashed line) is obtained by difference between the total and urban components. For O_3, the urban, total and background components for each city have been scaled so that they correspond to a 20% reduction from CLE levels (see text for further details).

the results for O_3, model responses were scaled assuming linearity so that for each city they correspond to a 20% emission reduction from CLE levels for NO_x and NMVOC. O_3 model responses were therefore corrected with NO_x and NMVOC-based emission coefficients for the MFR1 and MFR2 scenarios, respectively. PM model responses were not corrected since both the MFR1 and MFR2 emission reductions are built from a group of species which may potentially be contributing to PM production (see Table 1 for differences in PPM emission reductions across cities).

Along the East–West transect, the background impact is relatively homogeneous and leads to an average decrease of the O_3 concentrations of about 1–2 ppb for the MFR1 reduction (NO_x group), and of 0.1–0.5 ppb for the MFR2 scenario (NMVOC group).

The urban impact of MFR1 emission reductions indicates a clear increase in O_3 concentrations due to the reduced titration effects in the city area. With the exception of Berlin, this increase due to local measures is not counterbalanced by the background impact and the total impact remains "positive" in the CC (in the sense that a net increase of O_3 is still obtained). Air quality improvements outside the city are mainly due to regional control measures. NMVOC emission control (Fig. 6) shows a clear benefit in Milan, whereas for Berlin, Paris and Prague almost no impact is seen.

For $PM_{2.5}$ (annual mean) the impact on background concentrations of the MFR1 reductions is of the order of 1–2 $\mu g\,m^{-3}$ in all cities (Fig. 7), whereas it is of the order of 0.2 $\mu g\,m^{-3}$ for the MFR2 scenario (Fig. 8). In Prague and Berlin, the MFR1 background impact is of the same order of magnitude or slightly larger than the urban impact,

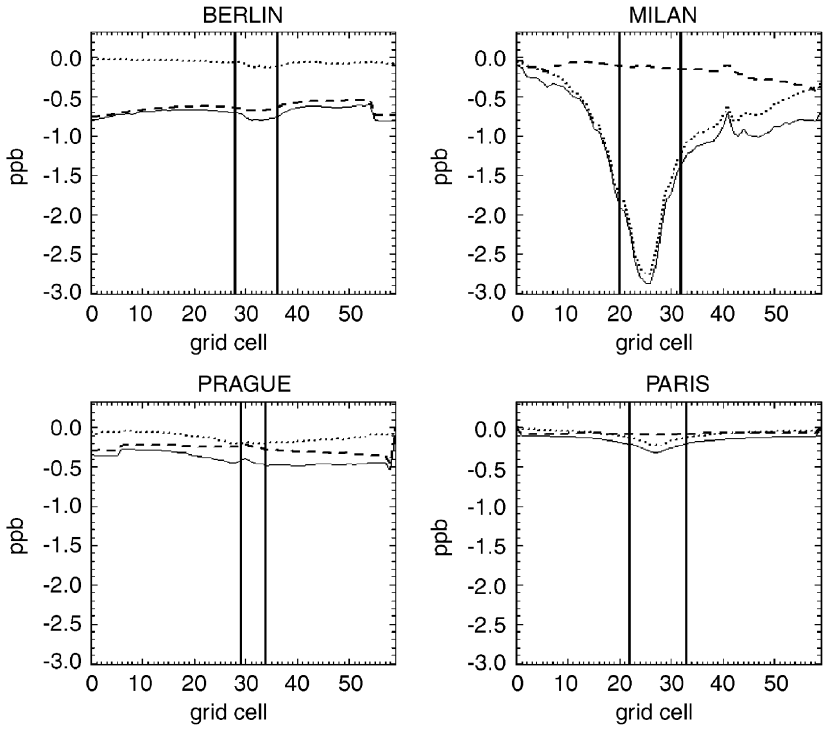

Fig. 6. Similar to Fig. 5, but for MFR2 (NMVOC, NH_3 and CO) emission reductions.

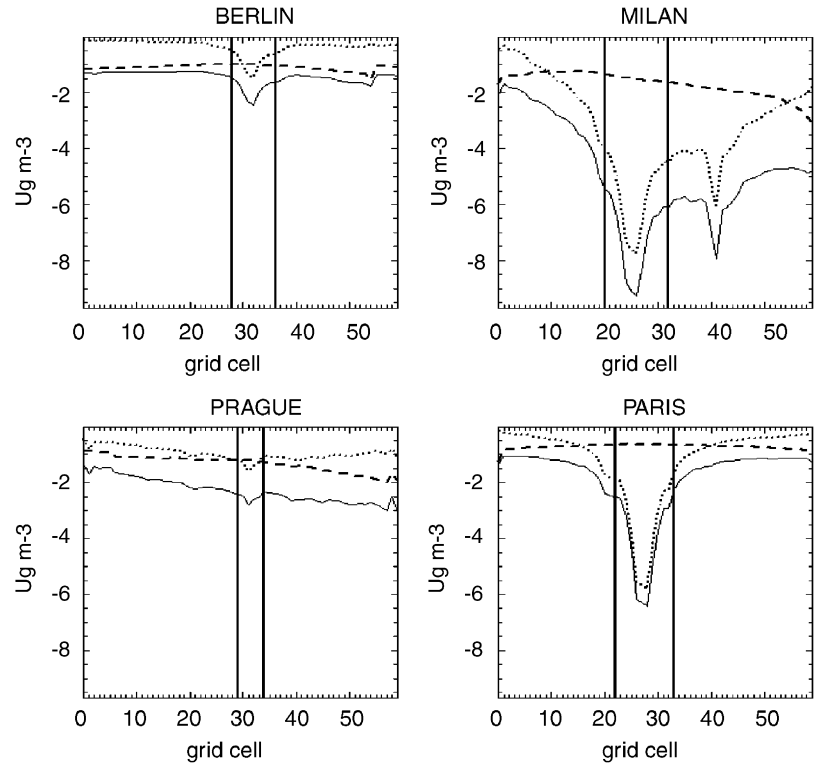

Fig. 7. Similar to Fig. 5 but for $PM_{2.5}$ yearly average.

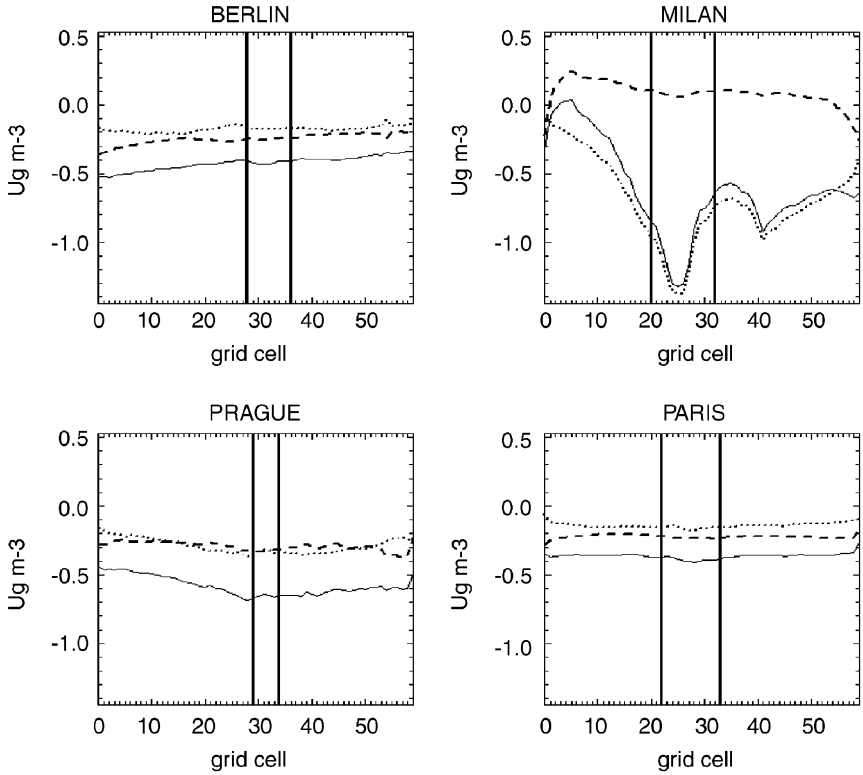

Fig. 8. Similar to Fig. 6 but for $PM_{2.5}$ yearly average.

whereas it is clear that for Milan and Paris local measures are more efficient. In the case of the MFR2 measures, Milan remains the only city where local measures are more efficient than regional ones. It must be noted that MFR1 measures are much more efficient than MFR2 in reducing ambient $PM_{2.5}$ concentrations, which may be explained by the fact that MFR1 emission reductions include PPM. At urban level, reductions range from 1 to $9\,\mu g\,m^{-3}$ for MFR1 against 0.5–$1.5\,\mu g\,m^{-3}$ for MFR2. The limited number of cities in this study prevents drawing general conclusions. Additional cities are necessary (currently under investigation) to evaluate the role of different factors, e.g. city size, meteorology, etc. on the efficiency of regional versus local measures to bring down PM and O_3 levels in urban areas.

6. Conclusions and discussion

The main aim of this paper has been to investigate how air quality models applied at different scales would predict pollution levels in response to emission control strategies in various cities in Europe. These models have been used in their own configuration related to meteorological inputs, biogenic emission parametrization and boundary conditions. The participating models generally agree on the O_3 changes expected from current legislation in 2010. There is agreement about less scope for further improvements related to scenarios emission controls beyond CLE. Regarding PM_{10}, larger differences between the models are observed. Benefits beyond 2010, due to MFR controls, are quite comparable to those obtained from CLE.

However, for the most restrictive scenario (MFR), the impact of emission reductions on O_3 and PM concentrations is of the same order of magnitude as the variability among models. This limits our ability to interpret the results in terms of scenario efficiency for scenarios beyond 2010. For O_3, the large variability among model results related to emission reduction scenarios may be explained by the chemical mechanism parametrizations which differ from one model to another and by the lack of homogeneity between model inputs (boundary conditions, meteorological parameters, etc.). Effect of chemical modelling concerns in particular the representation of the titration effect which strongly

depends on the NMVOC/NO_x ratio, and on the simulation of the responsible terminal reactions. In terms of emission control efficiency, NO_x reductions are more effective than NMVOC reductions to reduce SOMO35 levels over the CD.

In general, FS models are able to capture important urban scale effects, which are not represented by regional scale models. LS-models overpredict O_3 concentrations in the city area due to rough representation of the chemical mechanisms and they underpredict PM mean concentrations. In terms of responses to emission changes on O_3 mean levels and SOMO35, LS models tend to overestimate the impact of NO_x and NMVOC controls compared to FS model results. The impact of NO_x cuts on PM is generally underestimated by LS models, whereas responses to NMVOC cuts are quite similar both at LS and FS.

The series of emission scenarios designed to address the question of the efficiency of local emission controls towards regional measures, has led to city specific conclusions. While both regional and local measures have similar impacts on PM levels in Berlin and Prague, local measures are clearly much more efficient than regional ones to reduce pollution levels in Paris or Milan. For O_3, titration effects strongly hamper the NO_x control efficiency, resulting in an increase of O_3 in the CC. This is counterbalanced by the impact of regional measures only in Berlin while for other cities the net effect remains negative.

Finally, reliability of chemical transport model results strongly depends on the quality of the meteorological inputs. The variability associated with the results obtained for the city of Milan provides a clear example. This city is indeed characterized by very stable meteorological wintertime conditions which models do not capture well. Improvements in the representation of local meteorological conditions will increase the robustness of the model responses to emission controls.

As a conclusion, it should be noted that CityDelta results bring a new quantification of the differences between LS and FS air quality modelling. The importance of FS modelling has been demonstrated for O_3 and PM, for specific purposes related to urban centers. Variability of the model responses to emission reduction scenarios has been investigated. The first results obtained are quite consistent among cities. However, questions still remain considering that variability among results is of the same order of magnitude as the impacts of scenarios beyond 2010.

Based on a limited set of cities the conclusions derived from this second phase of CityDelta may not be generalized in a straightforward way. Future work is now necessary to increase the robustness of the conclusions. This is the subject of the third phase of the project in which additional cities and scenarios are planned to be incorporated.

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