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EFFECT OF SPATIAL AND TEMPORAL RESOLUTION ON EXPOSURE MODELING

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Abstract:Assessment of human exposure to air pollution is an essential part of environmental risk studies. As an illustration of the uncertainties and methodologies which can be used to assess the uncertainties we present case studies on the effect of spatial and temporal resolutions on the exposure modeling analyzing the sub-grid variability and its impact on city-scale exposure estimates. These results indicate that significant errors in the population weighted concentrations can occur due to the use of finite grid sizes. The methodology implemented in this study enables the improvement of estimates of the population weighted concentrations, which are used in long term health impact studies. The potential of the method is that low resolution models, can be used for fast multiple scenario or sensitivity calculations even as retaining their ability to calculate population exposure.The results of the small scale exposure modeling studies are very much in line with the large scale correction factor analyses although the small scale studies are based on one specific city area (Helsinki metropolitan Area) and one specific pollutant ($PM_{2.5}$), and in other cities and for other pollutants the exact values for correction factors will differ from this case study. However, we have demonstrated here a method for estimating these resolution effects in smaller scales, which can be readily utilized in any city and for every pollutant, up to the finest resolutions where emission data, population data and model calculations are available. We have also demonstrated that temporal resolution used in exposure calculations can have a major effect on the relative contributions of traffic, home and work locations to the total exposure. Especially the exposures in traffic and at work will typically be underestimated while the exposures at home will be overestimated.

Key words: *population weighted concentration, sub-grid variability*

INTRODUCTION

Assessment of human exposure to air pollution is an essential part of environmental risk studies. Exposure to air pollution can be estimated either by direct assessment using personal monitors or by modeling.Models for estimating exposure can be classified into three major groups (e.g. Ashmore and Dimitripoulou, 2009): estimates based simply on relating health aspects to modeled outdoor concentrations; empirical regression models; and mechanistic or mathematical modeling. The latter approach is the most complete but also more complex. The assessment of exposure with mechanistic approach usually requires application of integrated model chains starting from estimation of emissions, atmospheric dispersion and transformation of air pollutants. The exposure model then combines ambient air concentrations of pollutants and population activity data to calculate human exposure concentrations. An important concept in mechanistic exposure models is microenvironment, i.e. a location in which human exposure takes place such as home, school, workplace or car. The average personal exposure is estimated as a linear combination of concentrations in different microenvironments, weighted by the time spent in each of them.

Any modeling analysis has a number of limitations and uncertainties, which increase with the complexity of the integrated modeling system. When assessing human exposure there are several limitations to be considered, in particular the compromise made between spatial resolution and calculation domain. This limitation is of major importance since grid resolution may have an impact on the physical and chemical descriptions of the models. Additionally, combining ambient air concentrations of pollutants and population activity data with different resolutions will affect the exposure estimates. Since regional scale chemical transport models(CTM) do not capture the same spatial variability as the population, sub-grid variability (SGV) will also impact the exposure estimate. To deal with this question various schemes have been employed, e.g. CityDelta (Amann et al., 2007), that parameterize the ‘urban increment’. This represents the concentration difference between urban and regional areas and is employed to improve the population exposure estimates in urban areas. An alternative to implementing an ‘urban increment’ is to simply increase the model resolution to better represent the population variability, though this is highly impractical on continental scales for long term assessments.

Karvosenoja et al. (2010) assessed population exposure caused by the emissions of primary fine particulate matter (PM2.5) originated from road traffic and domestic wood combustion in Finland in 2000 and 2020.Their general implication was that the exposure values evaluated using integrated assessment models can be sensitive to the methodology, especially these can substantially increase with an increasing spatial resolution. It should not be necessary to increase the model resolution to resolve the concentration and population variance in an urban area and it should suffice to resolve the covariance of the population and concentration fields. In other words, it is not necessary to increase a models resolution to improve the population exposure estimate if the population or concentration fields are uncorrelated or homogenous, since enhancing the model resolution will not improve the population exposure estimate when that estimate is based on average concentration exposure.

The methodology implemented in this study accounts for the sub-grid variability of concentrations and their spatial correlation with population distribution. This enables improved estimates of the population weighted concentrations, which are used in long term health impact studies. The potential of the method is that large grid sizes, i.e. low resolution models, can be used for fast multiple scenario or sensitivity calculations whilst retaining their ability to calculate population exposure. The only requirement in regard to input data to the parameterisation is that emission data must be available at a suitably high resolution. Since the parameterisation includes the emission population covariance any changes in emission, or population, distributions in future scenarios will be implicitly included in the parameterisation. Additionally, we present an estimate of the error made when calculating the population weighted concentrations using typical CTM finite grid sizes.

METHODOLOGY

A study has been carried out to quantify the effect of sub-grid variability (SGV) of concentrations and population on exposure. The discretized population weighted concentration ($C_{pw,j}$) over any defined area Aj for a given period of time can be written as :

$$C_{pw,j} = C_j \left(1 + COV_{CP,j}\right) \quad (1)$$

where C_j is mean concentration for each grid square j and $COV_{cp,j}$ is the correction factor. $COV_{cp,j}$ was assessed and parameterised based on empirical data, by applying spatial statistical methods. This involves determination of the accumulated cross-variogram, which provides the covariance of two spatially distributed data fields, for a range of effective grid resolutions, or lag distances (d). This is calculated using Equation 2.

$$COV_{cp,d} = \frac{\sum_{i}^{n}\sum_{k\neq i}^{m}(c_i - c_k)(p_i - p_k)}{\frac{1}{2}\sum_{i}^{n}\sum_{k\neq i}^{m}(c_i + c_k)(p_i + p_k)} \qquad \text{for } k \in r_{ik} \leq d \quad (2)$$

where the index k goes through all stations located maximally at distance d from the measurement station location i.

Denby *et al* (2011) applied this methods to nitrogen dioxide (NO2), coarse particulate matter (PM10) and the ozone (O3) indicator SOMO35 using data from AirBase (AirBase, 2010) for the year 2006. Only regional and (sub)urban background stations have been used in the study and population data at a resolution of 3 x 3 km^2 is used as representative for these background stations.

To assess the correction factor for finer (<5 km) resolutions we have applied the EXPAND-FMI model for Helsinki area.EXPAND (EXPosure to Air pollution, especially to Nitrogen Dioxide and particulate matter) is a mathematical model for the determination of human exposure to ambient air pollution in an urban area (Kousa *et al*, 2002). The approach based on measurement data is no longer applicable in these resolutions as there are not enough measurement stations located on distances less than 5 km to provide any statistically significant cross-variogram data. Utilizing the exposure model with varying fine resolution (from 25 m to ~5 km) is a very straightforward way to obtain information on the cross-correlations of population and concentration data. However, as the calculations have been performed only utilizing Helsinki-area data, and for just for one specific case, further studies are still needed to ensure that the results can be generalized. It is obvious that at these scales the urban structure, especially the relative locations of the most trafficked areas and most densely populated is the decisive factor for cross correlation of population and concentration distributions and also on the effect of resolution on these correlations. in this case, the correction factor is obtained by estimating the total calculated exposure with varying resolutions and taking total exposure calculated at 46.4 km resolution as the reference exposure (correction factor = 0, for 6.4 km resolution)

RESULTS

The SGV and its impact on European scale (Denby *et al*, 2011) indicate that significant errors in the population weighted concentrations can occur due to the use of finite grid sizes: (i) the NO_2 $COV_{cp,j}$ is more strongly dependent on grid resolution than is the PM_{10} factor, probably due to the relatively high correlation between NO_2 concentrations and population density; (ii) the PM_{10} $COV_{cp,j}$ shows a weak dependence on grid resolution since PM_{10} concentrations are spatial homogeneous; (iii) SOMO35 shows a negative correlation, likely due to NOx titration in urban areas, and as such O_3 exposure estimates will be overestimated by 15% when finite grids of 50 km or more are used.

For finer scales (< 5 km), the correction factor as function of resolution seems to be smaller. In figure 1 the correction factor as a function of resolution, based on fine scale exposure model calculations for different microenvironments (home, work, traffic and other than previously stated) in the Helsinki Metropolitan area is presented.

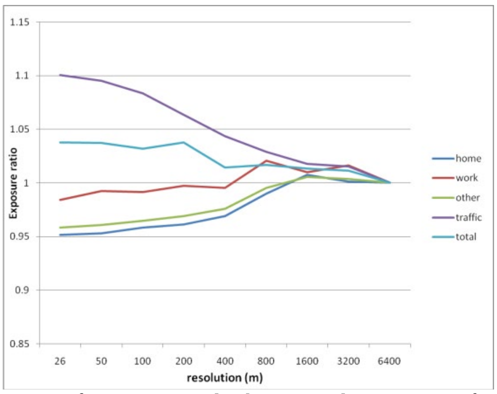

Figure 1. Correction factor for total exposures for PM2.5 at Helsinki Metropolitan Area as a function of effective grid resolution (m).

It can be seen that the correction factor for the total exposure is small (<0.05). For traffic exposures, where the effect of resolution is expected and observed to be highest for the cross-correlations, the maximum correction factor is ~10%. The results of the small scale exposure modeling studies seem to be very much in line with the large scale correction factor analyses, although, as already pointed out, the small scale studies are based on one specific city area and one specific pollutant, and it is to be expected that in other cities and for other pollutants the exact values for correction factors may differ from this case study. However, we have demonstrated here a simple method for estimating these resolution effects in smaller scales, which can be readily utilized in any city and for every pollutant, up to the finest resolutions where emission data, population data and model calculations are available.

The covariance correction factor (Equation 1) can be readily translated from spatial to time domain, showing that the correction factor for some specific location for exposure calculations with varying temporal resolutions is directly linked to the covariance of the time-series of concentrations and population activities (amount of population at some specific location). Similar methods for studying the effect of resolution to exposure estimates as demonstrated for spatialresolutions can be readily applied for sensitivity studies in time domain. To give a concrete idea on the effect of temporal resolution we utilize data from Helsinki Area 2008-2009 to illustrate the difference between exposures calculated with hourly vs. daily temporal resolution.

Figures 2 and 3 show the averaged diurnal variations of measured concentrations and activity distribution in Helsinki area, respectively.

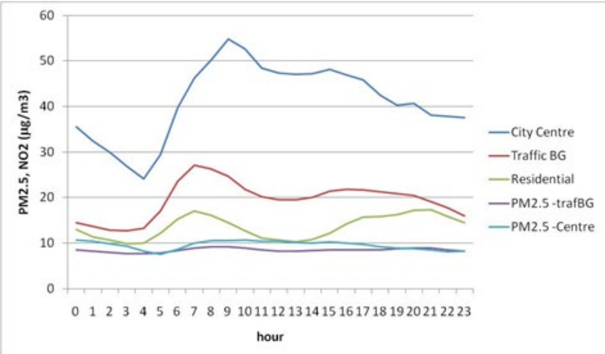

Figure 2. Hourly variation of measured concentrations of NO2 and PM2.5 in Helsinki in 2008-2009.

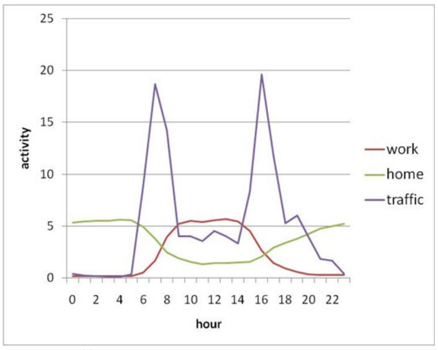

Figure 3. Diurnal variation of work , home and traffic activities in Helsinki area.(FMI-EXPAND exposure model, Kousa et al, 2002).

Figures 2 and 3 suggest that a strong cross-correlation between activity and concentration time series will occur for NO_2 concentrations while the temporal variations of $PM_{2.5}$ concentrations (Figure 2) are smoothed out by the dominating background concentrations. For NO_2, the cross-correlation of work and traffic activities are strongly and positively correlated with the diurnal variation of concentrations measured at downtown locations but negatively correlated with the diurnal variation of home activities (Table 1).

Table1. The relative change in exposures calculated using hourly vs. daily temporal resolution, by cross-correlating concentrations measured at downtown locations and activities

	work	home	traffic
centre	17 %	-8 %	12 %
traffic/bg	10 %	-6 %	17 %
residential	-6 %	0	9 %

This simple example clearlydemonstrates that, depending on temporal resolution used in exposure calculations, relative contributions of traffic, home and work locations to the total exposure can change very significantly. Especially the exposures in traffic and at work will typically be underestimated while the exposures at home will be overestimated.

CONCLUSION

As an illustration of the uncertainties and methodologies which can be used to assess the uncertainties we have presented analyses and case studies on the effect of spatial and temporal resolutions on the exposure modelling. The results from the analyses of the sub-grid variability and its impact on European wide and city-scale exposure estimates indicate that significant errors in the population weighted concentrations can occur due to the use of finite grid sizes. It is shown that the sub-grid covariance is the defining factor in determining this error.

The methodology can only provide information on the covariance down to the resolution of available data. Whilst high resolution data (< 1 x 1 km2) is available for population and altitude data for all of Europe, the current emission inventory resolution is significantly larger than this (~6 km). If the covariance is to be assessed at higher resolution then alternative data sources, e.g. land use or regional emission inventory data, would need to be used. Even so, the current calculations using monitoring data indicate that at grid resolutions of less than 10 km the covariance correction factor is less than 7% for all compounds.

For the high resolution case, traffic exposure is the most sensitive component, and the maximum correction factor for $PM_{2.5}$ exposures was found out to be 10%; for total exposure the correction factor was ~4 %. This lower sensitivity to resolution can be explained by the negative correlation of home locations and high-concentration areas, which counters the effect of stronger and opposite resolution dependence of traffic exposure. $PM_{2.5}$ exposures are expected to be much less sensitive to resolution changes, as the spatially smoothly distributed PM2.5 background concentrations dominate the exposures. Temporal resolution used in exposure calculations can have a major effect on the relative contributions of traffic, home and work locations to the total exposure, especially the exposures in traffic and at work will typically be underestimated while the exposures at home will be overestimated.

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