Estimating the contribution of commuting on exposure to particulate matter in European urban areas

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Front page picture
Snowstorms: the fear of many commuters. On 3 February 2012 an (unexpected) layer of snow of 10-14 cm resulted in a chaos in The Netherlands. Delayed and cancelled flights, more than 800 kilometre traffic jam, and until late in the evening no rail traffic in the western part of the country (photo: © Frank de Leeuw, 2012).

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Summary

The diversity of environments and lifestyles across Europe poses a challenge when it comes to estimate the impacts from the exposure of air pollution experienced by the population. A summary of previous case studies shows that improved spatial resolution (from 10x10 to 1x1 km) has small to moderate effects (estimated + 3%) on the calculated population weighted exposure to PM_{10} estimated using dispersion models.

For generalisation at the European level, the PM_{10} concentration variation within a city was analysed using data in AirBase. The results suggest that the urban and suburban background concentration levels typically vary within 15% (with a 95% confidence limit) around the city averaged background concentration. The PM_{10} concentration fluctuations at a station seem not to be correlated with the resident population density around the station. An analysis of the effect of taking into account the intra-urban variability, as given by measurements, in estimating the population exposure at the European level supports the finding from the case studies, namely, that the intra-urban variability has a small effect on the estimates. In preparing a total health impact assessment for the population in a city as a whole these concentration variations should not be considered when the presently used WHO linear dose-response function for PM_{10} is assumed, although the most exposed population groups have a larger health effect than the groups lesser exposed.

Using the WHO linear dose-response relationship for long term PM_{10} exposure, it is sufficient to take account of the commuting in terms of its effect on the average population exposure when considering the impact of air pollution on the total population. The present study indicates that the effect of commuting on the average exposure in European cities should be taken into account; it could be of the order of 5%, varying from 0 and up to 10% or more for individual cities. Taking into account also the exposure during the commuting will increase this added exposure further.

The exposure experienced by the fraction of the population at the high end of the exposure distribution will lead to a higher risk for this group. Commuters form a high-exposed group both because of their moving to work or school in the central parts of the city where concentrations in general tend to be higher, as well as because of exposure during the travel itself. The Oslo case study gave as an average for 19 test persons, a 52% increase with a maximum of 160%. At the general population level an increase of 0 – 12% was found in the other case studies.

PM_{10} concentrations show a diurnal profile. When urban concentrations are larger during working hours than the annual mean values, the exposure of commuters will be increased. Using information from AirBase, factors for day-time and night-time concentrations have been determined as basis for day-time and night-time concentration maps. This information, together with generalized information on commuter flux into and out of a city has been introduced in a simplified commuter model. At the moment, lack of detailed information on commuting in European cities precludes a more enhanced approach. Using this simplified methodology the effect of commuting (the shifting of position from home to work/school) on the population weighted mean is estimates at 5 to 12%, in good agreement with the results from the previous case studies. However, for the individual commuter increases of 50% to more than 300% can be expected, when the exposure during the travel itself is also added, as also shown by the Oslo case study.

Thus, the fraction of the population exposed to the highest concentrations, the high end of the exposure distribution, is increased substantially by the commuters, their number is large and their increased exposure substantial. If it becomes important for health effects reduction to limit the high end of the exposure, it is important to further investigate the size of the high-end population, their actual exposure, and to include commuting and commuters in the exposure assessment.

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1. Introduction

1.1 Extraction of results from work in 2008 and 2009

The starting point of this work is the need to explore to what extent current methods used by ETC/ACM to assess the exposure of the European population to air pollution describe the actual exposure well enough, and how the methods could be improved. The exposure assessments are used as a basis for estimating the impacts of the air pollution on the health of the European population. If the exposure assessment can be questioned, the same is true for the health effects estimates resulting from it.

Assessments of the impacts of population exposure to air pollution upon the health of European citizens are currently performed by the Topic Centre based on 1) average air concentrations at 10 km x 10 km scale, and 2) static residential populations. Whilst this generalisation is sufficient for broad assessments, the question arises whether some factors influencing population exposure significantly, potentially leading to systematic underestimation of exposure, may be overlooked.

The current estimates are essentially derived from monitored air quality data combined with results from air pollution modelling. The relatively large spatial resolution of 10x10 km, with consequent use of grid cell average concentrations without considering sub-grid intra-urban gradients either in concentrations or population densities raises uncertainty as to the accuracy of assessments. Smaller cities will not be resolved within this resolution. The low resolution results in a systematic underestimation of exposure at the high end of the exposure distribution, given the correlation that exists between population density and air pollution concentrations. The gradients in population density and air quality in cities may be large. Also, populations move daily across zones of different pollution levels as well as commuting from low to higher polluted areas along highly polluted road corridors. The scale of averaging can soften the gradients that actually exist.

The underlying theme is to assess the potential importance of actual exposure patterns as compared to the current static 10x10 km approximation. Even if the ‘dose’ in dose-response relationships is to a large extent based on proxies for exposure, like results of stationary measurements, the space-time details of individual exposures could be utilised for more effective exposure-reduction measures.

The main objective of this work is to investigate population exposure methodologies, their uncertainties, systematic shortcomings and potential improvements, concentrating on:

- how smaller scale resolution can improve exposure estimates;
- how the mobility of the population (commuting) affects exposure; and
- how commuting can be taken into account in improved methodologies for exposure assessments on the European scale.

Most of the population in Europe lives in urban areas. Thus the studies have concentrated on the urban scale, although it is also important to take into account the exposure of the population living in more rural areas. It has been established that exposure to particulate matter (PM) and ozone in ambient air are the main reasons for air pollution related health effects (see, for example, WHO 2006, 2008). Quantifying the health impacts in terms of premature deaths shows that at current levels, the largest impact on health must be attributed to PM exposure (see for example, EC 2005).

This study was conducted over the years 2008, 2009 and 2010, with various sub-objectives and focus between the years. It has included a review of air pollution exposure methodologies, as well as case studies using data for the cities of London, Athens and Oslo as well as for the Moravia-Silesia industrial region (Barrett et al. 2008; 2009). Results from the work during 2008 and 2009 are summarised below.

Literature review on exposure modelling and recent results

The study (Barrett et al. 2009) has given an overview of the state of the art on exposure science and recent results on improved air pollution exposure assessments at the urban scale. The main focus has been on those studies dealing with factors such as spatial scale and commuting. The variety of exposure models is large and the combination of different models to improve the assessment results increases the number of possibilities and the complexity.

Given the current state of the knowledge, hybrid models where the strengths of different methods are combined constitute an important tool for reducing uncertainties in exposure estimates. The hybrid models use a combination of models based on principles of physics and chemistry (Mechanistic models) and models based on statistical relationships (Empirical models). The variability of the concentration field at urban scale has been intensively studied. The uncertainties associated with the methods and the variability observed at urban scale represent important challenges for performing accurate exposure assessment.

The importance of human behaviour (human activity) and its effect on the exposure estimates and therefore on the health impact assessment have been established in several studies. Current exposure assessment estimates are commonly based on static residential address which may involve high uncertainties. An increase of 50 and 100% in the exposure has been considered as reasonable in some studies when taking into consideration commuting in the exposure modelling (Pérez Ballesta *et al.* 2008; Barrett *et al.* 2008). The impacts of various microenvironments on the personal exposure have been investigated by many studies (Barrett et al., 2009). A good example of this type of work is the EXPOLIS study (http://www.ktl.fi/expolis/files/final_report.pdf). The case studies and the literature review points to the need for additional research on exposure assessment methodologies, and comparison of estimates from different exposure methodologies may contribute to the better understanding of exposure estimate uncertainties.

Due to commuting between home and work place, the population number within cities undergoes large variations. During working hours a doubling or even tripling of the population for city centres have been suggested by Borrego *et al.* (2006) and Barrett *et al.* (2008). Similarly 70% increase has been reported for some American cities, whereas population increases up to 40% have been observed from the available data from Dutch cities (Table 1.1). These population shifts may involve important changes in exposure estimates compared to those static-based estimates. Thus information about the time that people spend in the different microenvironments (e.g. transport, home and work) constitute an important key to reduce uncertainties on exposure models and therefore to improve health impact assessment. Typically, people spend 20-30% of their time away from their home, and typically 4-5% on the commuting itself with a typical commuting journey length of 0.7-1.7 hours per day (Kousa *et al.* 2002; EC 2004; OECD 2008). How to deal with time-activity information varies between studies, some of them consider data from time-microenvironment-activity (TMA) diaries (Kousa et al. 2002; Perez Ballesta et al. 2008)) or dynamic transport models (Beckx et al. 2009), others follow selected or example individuals, compare static home population and static work population, whereas others consider static home population with statistical description of the variations (Barrett et al. 2008). More research is needed on the analysis of mobility or time-activity-space patterns.

Table 1.1: Population increase due to commuting in American cities with different population densities (U.S. Census 2000) (Left) and Dutch cities (Vos and Trijssenaar 2000) (Right).

Location	Population Increase (%)
Population over 1 million	
Houston	20,64
Dallas	19,15
San Diego	11,60
Population between 500 000 - 999 999	
Washington D.C.	71,8
Boston	41,1
Seattle	28,4
Denver	28,0
Portland, OR	23,0
San Francisco	21,7
Charlotte, NC	21,2
Houston	20,6
Nashville	19,5
Austin	19,4
Population between 250 000 - 499 999	
Atlanta	62,42
Tampa	47,47
Pittsburgh	41,30
Population between 100 000 - 249 999	
Irvine	73,83
Salt Lake City city	72,18
Orlando	70,72

Location	Population Increase (%)
Haarlemmermeer	44,08
Utrecht	31,57
Arnhem	29,35
s-Hertogenbosh	21,63
Eindhoven	20,23
Maastricht	18,74
Groningen	16,67
Amsterdam	16,00
Breda	13,72
Rotterdam	12,21
Nijmegen	11,80
Apeldoorm	11,25
s-Gravenhage	7,92
Enschede	6,26
Amersfoort	5,64
Tilburg	5,15
Dordrecht	1,58

Results from the case studies: Increased spatial resolution

The effect of increased spatial resolution on the population exposure estimate has been investigated in the case studies of Moravian-Silesian area and in London (Barrett et al. 2009). Dispersion models were applied in the two areas, and the effect of increasing the resolution from 10x10 km to 1x1 km was calculated. The population was considered stationary (no commuting or other systematic movements) in these calculations, so the results reflect only the improved spatial resolution.

The main results were as follows, moving from 10x10 km to 1x1 km:

- In Moravian-Silesian area, the population weighted average exposure (PWE) to PM_{10} increased from 39.9 µg/m^3 to 41.2 µg/m^3, or by 3.3% (Table 1.2).
- In London, the population weighted average exposure (PWE) to PM_{10} increased from 23.1 µg/m^3 to 23.8 µg/m^3, or by 2.8%.

Thus, the two case studies gave a rather modest stationary population exposure increase when improving the spatial resolution. The two case study areas are quite different. Moravia-Silesia is a rural-township-urban industrial area in central Europe with a population of about 1.9 million people. Rural concentrations are not much lower than urban concentrations due to small scale heating with coal and wood in rural and township dwellings, leading also to typically higher evening-night-time than daytime PM concentrations in the area. London is a very large urban region with a population of more than 7 million people and an emission structure dominated by road vehicles. The two case study areas are quite different; although the effect of the improved spatial model resolution is similar in the two areas, the result is not general for all areas..

The spatial resolution effect has also been studied on the European scale under another ETC/ACM activity (Horálek et al. 2010). At the European scale the increase in resolution from 10x10 km to 1x1 km results in an increase in population weighted annual PM10 concentration from 27.1 to 28.6 $\mu g/m^3$, or by 5.5% (Horalek et al. 2010).

These average percentage increases hide the underlying structure in the population exposure distributions. The Moravian-Silesian case study provided results shown in Table 1.2. The largest increase in exposure brackets was estimated for the highest concentration brackets, above 40 $\mu g/m^3$, while a smaller fraction of the population is exposed to the lower concentration classes. Thus, the increased spatial resolution showed that more people are actually being exposed to the highest concentrations. This is potentially a significant result. Estimating a health outcome, for example the number of premature deaths, the improved resolution will result in a 3.3 to 4.3% increase depending on whether a no-effect level of 0 or 10 $\mu g/m^3$ is assumed. However, while these differences at the population level might be considered to be minor, the results in Table 1.2 indicate that an increased fraction of the population is at a much higher risk than the population on the average risk: the fraction in the concentration classes of 40 $\mu g/m^3$ or higher increases with 7.1%.

Table 1.2: Population exposure to PM_{10} annual average concentration in 2006 in Moravian-Silesian Region.

Grid size [km x km]		*Concentration classes [$\mu g.m^{-3}$]*						*Population weighted PM_{10} annual average concentration*
		0-10	*10-20*	*20-30*	*30-40*	*40-45*	*>45*	
1x1	Number:	**0**	**12 911**	**189 653**	**717 067**	**320 565**	**620 920**	41.2 $\mu g.m^{-3}$
	% of pop.:	**0.0%**	**0.7%**	**10.2%**	**38.5%**	**17.2%**	**33.4%**	
10x10	Number:	**0**	**12 007**	**234 325**	**805,756**	**228 339**	**580 689**	39.9 $\mu g.m^{-3}$
	% of pop.:	**0.0%**	**0.6%**	**12.6%**	**43.3%**	**12.3%**	**31.2%**	

Results from the case studies: Including commuting in the exposure calculations

Commuting implies two central departures from static residential exposure: a) spending time at work, school, etc. and b) exposure during commuting journeys. The importance of these is determined by period and the magnitude of concentration changes due to travelling.

The case studies in London, Athens and Moravia-Silesia calculated the changes in population exposure due to point a) above, while the Oslo case study attempted to take account of both a) and b) above, thus also including exposure during the travel itself.

The main results were as follows:

- For London, the population weighted average exposure to PM_{10} (PWE, annual average), increased from 23.8 $\mu g/m^3$ to 24.4 $\mu g/m^3$, that is, by 2.5%.

- For Athens, the similar PWE increase was from 29.6 $\mu g/m^3$ to 30.9 $\mu g/m^3$, that is, by 4.4%, for the Athens-Pireaus area.

- For Moravia-Silesia, taking account of the time at work/school gave almost no change in the average PWE. This can be explained by the situation that although the daytime concentrations are higher in the central areas than in the rural areas, the daytime concentrations are generally lower than the rural night-time concentrations, and population groups move to central areas in daytime.

- For Oslo, it could be estimated that when taking account of outside-city centre dwellers commuting to Oslo central areas for work, and accounting for the estimated exposure during the commuting itself, the resulting increase in the mean concentration weighted over total population in Oslo is 6-12% (corresponding to an increase in PWE of about 1-1.5 $\mu g/m^3$),dependent upon assumptions. The commuting and travel exposure resulted in a 12-14% increase of the population fraction exposed over 20 $\mu g/m^3$. Dwelling areas near Oslo have generally rather low PM_{10} concentrations. For the commuters themselves (represented by 19 selected 'persons'), the concentration with daily routine (two hour travelling, 8 hours working in central Oslo) is on average 52% higher than the concentration at the home address. Large variations (from -12 up to +160%) are seen for the 19 test persons.

From the case studies, it is clear that commuting to and spending time at work would most often lead to increased average population exposure (PWE) to PM_{10}, although Moravia-Silesia does represent a case where the calculations do not show an increase. The magnitude of the increase clearly depends upon the local conditions, including the difference between central urban and suburban-rural area concentrations. Climate-meteorological conditions and their variation between day and night also play a role: day-night temperature and dispersion conditions modulate the concentrations caused by the differing emission strengths day-night. The Moravia-Silesia example shows that this can influence the commuting effect on the PWE significantly.

Incompleteness of the results of the study so far

It is clear that spatial resolution and commuting influence the estimated level of population exposure, although the case studies so far indicate that the influence may not be very large. Spatial resolution from 10x10 to 1x1 km increased the average population exposure by 2.8% to 3.3% in the two case studies (London and Moravia-Silesia, respectively). Commuting (spending time at work/school) increased the population exposure from almost nothing to more than 6% in the case studies (Moravia-Silesia, London, Athens, Oslo). The exposure during the commuting journeys has so far not been taken into account except for the Oslo case study, where it was done in a fairly simplified way. The example in Oslo showed a significant increase in the estimated population averaged exposure if exposure during travel was taken into account.

Another important issue is that although the influence of the mentioned factors may be limited when calculated as average over the entire population, it is clear that the commuting affects a certain and defined population group which would tend to have their exposure increased substantially compared to the non-commuting population group. Providing data for the population exposure <u>distribution</u> will show the magnitude of the population that is exposed to the highest concentrations. The most highly exposed group could consist partly of commuters, and partly of the group of urban dwellers who live and work in the most highly polluted areas. It would be of interest to have an estimate of the magnitude of the population group in Europe that is most highly exposed, and what the health impact of this high exposure actually is.

1.2 Exposure assessment methodologies in the Clean Air for Europe programme

Population exposure and health effects assessments of $PM_{2.5}$ are part of the basis for the development of the Thematic Strategy of the Clean Air for Europe (CAFE) Programme (http://europa.eu/legislation_summaries/environment/air_pollution/l28159_en.htm).

The CAFE assessments of population exposure to $PM_{2.5}$ and the resulting health effects assessments are based upon the following methods for exposure and health effects estimation (Amann et al. 2005b):

- Average background concentrations in the EMEP grid (50x50 km^2 grid size) is provided by the EMEP unified (eulerian) model.
- The model considers primary PM emissions as well as inorganic PM formation from anthropogenic emissions of SO_2, NO_X and NH_3.
  Natural sources of PM as well as organic secondary particles (SOA) are not considered.
- The increase in urban PM concentrations is estimated using the 'City-Delta' procedure (Amann et al., 2005a).
- The estimation of health effects by $PM_{2.5}$ is based upon the linear dose-response relationship recommended World Health Organization/UNECE Task Force on Health (http://www.unece.org/env/documents/2004/eb/wg1/eb.air.wg1.2004.11.e.pdf). It 'transfers the rate of relative risk for $PM_{2.5}$ identified by Pope *et al.* (2002) for 500.000 individuals in the United States to the European situation and calculates mortality for the population older than 30 years'.

The 'City-Delta' procedure is developed based upon the City delta study where 17 models participated in modelling air pollution in 8 cities for 7 different scenarios (Cuvelier et al., 2005). The functional relationship used to calculate the additional ('delta') PM concentrations 'relate the difference between the annual average $PM_{2.5}$ concentration in the urban area and in the 50x50 km grid cell surrounding the city with spatial differences in emission densities of low level sources and city-specific wind speeds' (Amann et al. 2007). Emission densities in grids and cities are related to population densities. Input to the urban increment calculations are EMEP grid cell emission densities, population density ratios and grid cell/city average wind speeds. The relationship is 'calibrated' using 'parameters derived from the City-Delta ensemble model'.

The urban population exposure assessment of the CAFE Programme thus uses a spatially uniform $PM_{2.5}$ concentration in each city. Also, the assessment considers the resident population, and does not consider the effects on average urban exposure of the commuting population. In response to this, the present study attempts to include the effects on urban population exposure when taking into account intra-urban variability and commuting.

Regarding health effects and its valuation, CAFE considered total mortality. It could be recommended to use cause-specific risk data which are regarded as more reliable, see section 1.3 below.

1.3 Relevant health endpoints

In ambient air, especially in urban air, a wide range of pollutants are found in relative high concentrations: nitrogen dioxide, ozone, particulate matter and on a more local scale, high levels of benzene, PAH or heavy metals are observed (Mol et al. 2010; Vixseboxse and de Leeuw 2009; Barret et al. 2008, 2009). This pollution originates from several sources, like traffic, space heating, industrial sources (see e.g. Moussiopoulos et al. 2008). Several studies (e.g. Krzyzanowski et al. 2005) have documented that in particular the exposure to traffic-related air pollution is associated with adverse health effects. According to current views, the largest risk for human health is attributed to particulate matter exposure. When expressing health effects attributable to the exposure to air pollution in terms of years of life lost or premature deaths, the exposure to PM is several times more important than the exposure to ozone or other pollutants. A comparison of mortality and morbidity health end-points is only possible when it is based upon economic valuation. Although it should be kept in mind that the basis of evidence for quantifying morbidity end-points is more limited than for mortality, cost benefit analysis as for example made in the CAFE process clearly shows that costs are dominated by mortality (Watkiss et al. 2005).

There is growing evidence that the fine PM fraction ($PM_{2.5}$) is more toxic than the coarse fraction PM_{10} minus $PM_{2.5}$). An evaluation on the basis of $PM_{2.5}$ is to be preferred, but for the purposes of this study the available information on measured ambient $PM_{2.5}$ concentrations on the European scale is insufficient. Atmospheric transport models have been used to calculate $PM_{2.5}$ concentrations but the spatial resolution is too coarse to resolve the urban scale needed to account for commuting and journeys. De Leeuw and Horalek (2009) have prepared a European $PM_{2.5}$ map with a resolution of 10x10 km. This map, prepared for 2005, is however based on PM_{10} measurements and an assumed $PM_{2.5}/PM_{10}$ concentration ratio. At the moment no detailed $PM_{2.5}$ maps on such a low spatial scale are available for the whole of Europe. Consequently, for the European scale this study will base exposure estimates and sub-sequent health effects assessment on the routinely available PM_{10} maps as prepared by the ETC/ACC (see de Smet et al. (2009) and references cited therein for more details). When $PM_{2.5}$ maps become routinely available, it is recommended to use, instead of the total mortality, cause-specific mortalities (see Table 1.3) as health outcomes.

The relative risks associated with a 10 μg/m^3 change in PM_{10} concentration are given in Table 1.3. Künzli et al. (2000) assume a linear concentration-response function over a concentration range of 7.5 μg/m^3 to 40 μg/m^3 and above. We make the assumption that the function can also be applied for the population fraction at the upper end of the concentration distribution. For completeness the relative risk for $PM_{2.5}$ are included.

Table 1.3. Mortality relative risk associated with a 10 μg/m^3 change in PM concentration.

pollutant	Health outcome	Relative risk or cases per 10 μg/m^3 (95% confidence interval)	reference
PM_{10}	Total mortality, adults > 30 year; excluding violent death	1.04 (1.03 – 1.06)	Künzli et al., 2000
$PM_{2.5}$	Total mortality, adults > 30 year; excluding violent death	1.06 (1.02 – 1.10)	Pope et al. 2002
$PM_{2.5}$	Mortality from cardio-pulmonary disease, adults > 30 year (ICD-9: 401-440 & 460-519)	1.08 (1.02-1.14)	Pope et al. 2002
$PM_{2.5}$	Mortality for lung cancer, adults > 30 year (ICD-9 162)	1.13 (1.04 – 1.22)	Pope et al. 2002
PM_{10}	Mortality from acute respiratory infection, children aged 0-4 years	1.01 (0.99-1.03)	Cohen et al. 2004

1.4 Objectives of the present study

The main objective of this study is to attempt to provide a stronger foundation for selection of policy measures to reduce population exposure to air pollutants and resulting health effects. The study bases itself on the previous work under the relevant ETC/ACC activities, which includes carrying out 4 case studies for London, Athens, Moravia-Silesia and Oslo (Barrett et al., 2008; 2009), summarised above (section 1.1).

Focus is on PM since, as detailed above, the health effects of PM dominates the costs of the population exposure to air pollution in Europe.

Population exposure as calculated under the CAFE work uses a uniform concentration of PM in each city, and does not take account of the day-time moving of population groups into the cities due to the population's commuting. The case studies indicated that the effect of taking account of the intra-urban concentration variability and increasing the spatial resolution to pick up the variability at enough detail, has a small/moderate effect on the population weighted exposure (PWE). They also showed, however, that taking account of the commuting can increase the PWE substantially, especially at the high end of the exposure distribution.

The more detailed objectives of the study presented in this report where to investigate:

- The exposure distribution and the most highly exposed part of the population.
- The contribution from commuting and journeys to the exposure.

The unique commuting data base of London to study the commuting effects on exposure, including the additional exposure during travel itself will be used in a subsequent study. This study is focusing on the following:

1. AirBase data analysis: investigate the intra-urban variability.
2. Examination of data bases on population and commuting data, to investigate the extent and completeness of European commuting data bases and how they can be utilised for exposure studies.
3. The European air pollution exposure situation resulting from the above activities.

Activities 1, 2 and 3 are described respectively in Chapter 2, 3 and 4.

2 Study of AirBase data relevant for assessment of PM_{10} exposure distribution: Intra-urban variability

When quantifying air pollution health effects and their distribution among the population, it is important to have information about the spatial variability of ambient concentrations as well as of the population. Using information available from AirBase we have explored the spatial variability of PM_{10} in urban areas. Using the data for 2008, 17 cities included in the Urban Audit dataset[1] have been selected where 4 or more (sub)urban background stations are operational. As we focus in this report on the impact of PM exposure to air pollution, and the health effects are dominated by the long term exposure, only the annual mean concentrations are analysed.

To evaluate the homogeneity of PM_{10} levels within a city the concentration at each of the monitoring stations in 17 cities has been compared with the city averaged concentration calculated over all operational stations. Figure 2.1 shows the relative deviation for each of the stations from the city average. In a few cases the maximum deviations are ±25%; generally the highest and lowest observed concentration is within ±15% of the averaged. The observed variability does not seem to depend on the size of the city: the relative standard deviation (= σ/C_{aver}) is not correlated with the size of the city population. There is also no indication that the variability correlates with the PM_{10} concentration.

A similar analysis was made starting with agglomerations as defined under the Air Quality Directive. For the 37 agglomerations with four or more (sub)urban background stations, the relative standard deviation ranges from 3% to 30%. This is, not unexpectedly, (slightly) larger than found for the Urban Audit cities, as an agglomeration frequently consists of several cities including the less populated areas between the cities a larger variation is expected. Again no correlation between the size of an agglomeration (expressed by the population number) and variability is found.

According to the Air Quality Directive 2008/50/EC sampling points directed at the protection of human health shall be sited in such a way as to provide data on the following:

— the areas within zones and agglomerations where the highest concentrations occur to which the population is likely to be directly or indirectly exposed for a period which is significant in relation to the averaging period of the limit value(s);

— levels in other areas within the zones and agglomerations which are representative of the exposure of the general population.

Assuming that cities actually locate monitoring stations according to these criteria, we may further assume that the monitoring stations are spatially representative for the urban domain, and we may conclude that the relative variability of PM_{10} concentrations in urban areas is typically within ±15% independent of variables like city size or current PM_{10} levels.

[1] http://epp.eurostat.ec.europa.eu/portal/page/portal/region_cities/city_urban

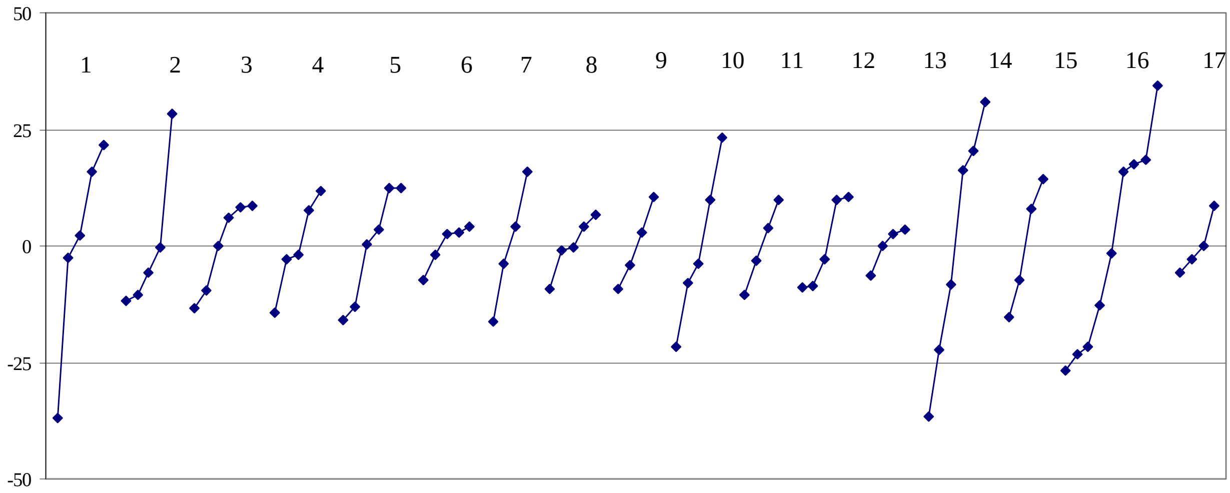

Figure 2.1. Variability of PM concentration in urban areas (2008): relative deviation at each individual station compared to the urban averaged concentration. Numbering and further details of the selected cities are given in Table 2.1

Table 2.1. Numbering, population averaged concentration (µg/m³) and standard deviation (µg/m³) in urban areas; PM_{10} annual mean for 2008.

number	name	population	conc	st.dev.	number	name	population	conc	st.dev.
1	Graz	235477	28.5	6.5	9	Marseille	1014110	30.3	2.6
2	Antwerpen	455148	22.6	3.7	10	Dublin	471841	13.7	2.4
3	Praha	1170571	22.3	2.1	11	Roma	2553873	30.7	2.7
4	Berlin	3387828	22.9	2.3	12	Warszawa	1692854	28.8	2.8
5	Hamburg	1734830	20.9	2.6	13	Poznan	570778	29.3	1.3
6	Toulouse	636245	19.5	0.9	14	Gdansk	459072	18.5	4.9
7	Bordeaux	700027	20.2	2.7	15	Lublin	355998	27.3	3.7
8	Lille	1098606	29.4	1.8	16	Istanbul	9897599	59.2	13.2
					17	London	7429200	23.5	1.4

As a first order approximation the concentration might be described by a normal distribution with a σ = 7.5%. This, however, relates to the spatial PM_{10} variability, and is not necessarily true for population exposure. At spatial scales of tens of kilometres or larger there is a positive correlation between population density and PM_{10} concentrations. If such a correlation also exists within an urban area, the population weighted concentration will exceed the spatially weighted concentration. To examine this we related the resident population density within a radius of 1 km around the station with the concentration. Before correlating, population density and concentration data are normalized according to:

$$dev_{ik} = (X_{ik} - \overline{X_k}) / \overline{X_k}$$

where dev_{ik} is the relative deviation at station *i* in city *k*; X_{ik} is the observed value (population density, concentration) at station *i* in city *k* and $\overline{X_k}$ is the averaged value in city *k*. Results are shown in Figure 2.2; there is no correlation between the relative deviation from the urban average for concentration and resident population density. For the small spatial scales within a city the correlation between resident population and concentration is obviously lost. Reasons can be the long-range transport character of PM pollution which results in a high urban background level from outside sources, as well as the typically complex spatial source distribution within cities, where sources a bit further away from the station location may influence the concentration significantly.

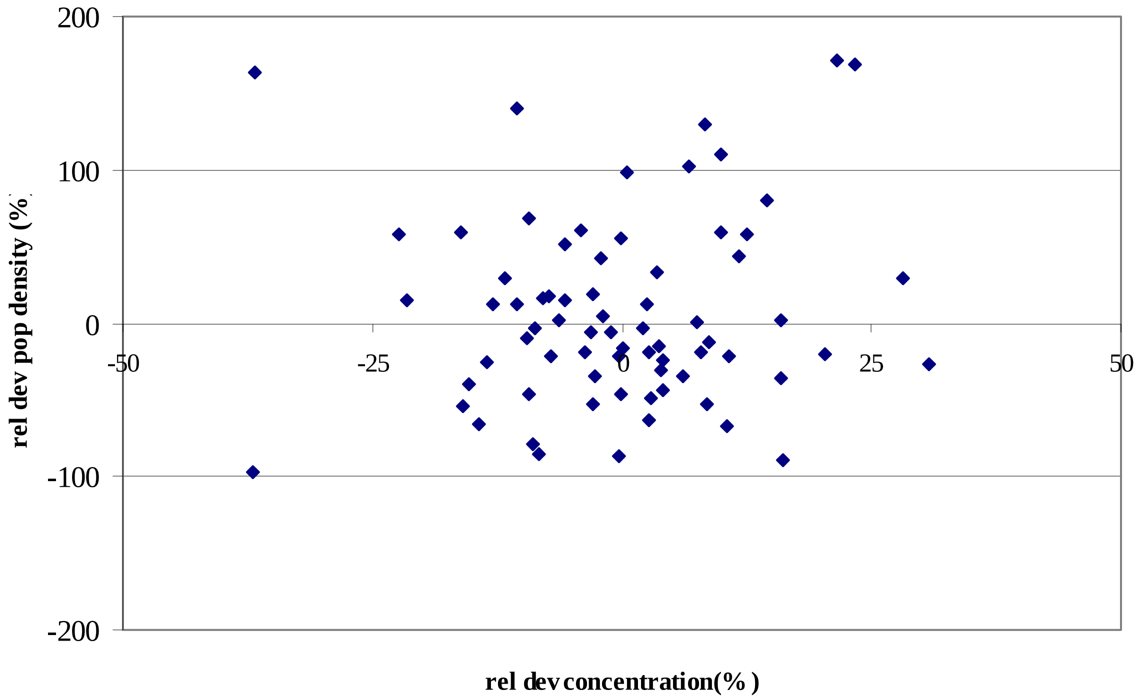

Figure 2.2. Relative deviation of population density around a monitoring station versus the relative deviation of the concentration at the station.

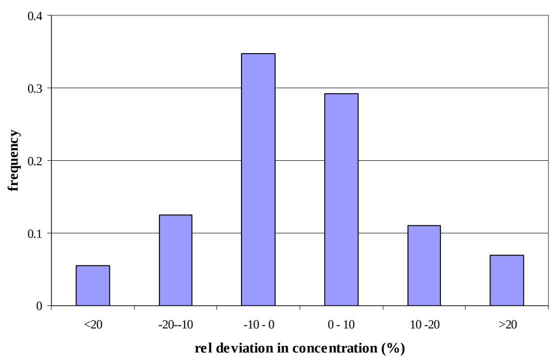

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Figure 2.3. Frequency distribution of the relative deviation of the concentration at urban background stations from the average concentration in each city.

A frequency distribution of the relative deviations in concentrations is given in Figure 2.3. Evaluating Figures 2.2 and 2.3, one may assume for the population exposure within a city a normal distribution with σ = 7.5%. When assessing the health impacts on the total urban population using linear dose-effect relations the concentration variability might be accounted for but its impact will not be large. For a linear dose-effect relation the incremental effect is independent upon the actual dose level. However, for non-linear relations or in case of a threshold level in the same order of magnitude as the current urban levels, the upper end of the population exposure distribution has to be considered. The above result that within the city there is no correlation between population density and concentration levels is based on a static (resident) population. The effect of a commuting subset of population is not accounted for. At the moment we have no information whether there is a correlation between total daytime population and pollutant concentrations. For European cities industry is generally located outside the city boundaries while working places are mostly found in commercial areas. Commercial areas will attract more traffic than residential areas. This leads to increased traffic emissions and ambient concentrations although the traffic contribution to concentrations may be too small to detect. As a first approximation we assume that concentrations in commercial and residential areas are similar and that commuters are exposed to a similar concentration distribution as the static population.

A frequency distribution for the European urban population exposure can be made from the information in the Urban Air Quality Indicator, CSI004 (EEA 2010). The 2008 data set of the Urban Air Quality Indicator contains 300 cities with a total population of 110.5 million. In Figure 2.4 two distributions have been calculated; one is based on assigning the total population in a city to the concentration class corresponding to the city averaged concentrations. The second distribution is based on first estimating a distribution within each city using σ = 7.5%; in a second step these distributions have been summed over all cities. Turkish cities have not been included in the analysis. The difference between the two distributions is small. The fraction of the urban population exposed to concentrations above the limit value of 40 µg/m^3 is about the same for both procedures: 9.8%.

The analysis above indicates that taking account of the intra-urban concentration variability at the spatial scale of the typical distance between monitoring stations in the cities, which is several kilometres, has negligible effect on the European average population exposure. The previous case studies gave as a result that increasing the spatial resolution from 10x10 to 1x1 km increased the modelled population weighted exposure (PWE) by about 3%. In any case, both these approaches indicate that the effect of intra-urban variability on PWE in modest, when commuting is not taken into account.

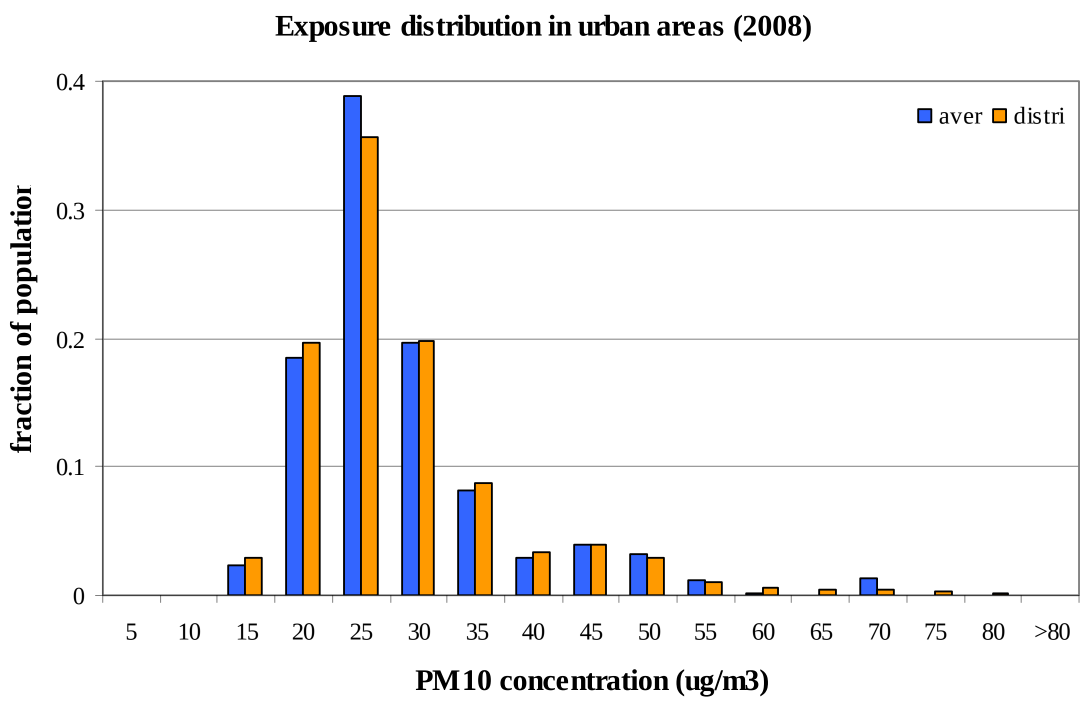

Figure 2.4. Frequency distribution of population exposure in urban area assuming no concentrations variations within a city (blue bars) and assuming a normal distribution with σ = 7.5% (orange bars). City information (population, concentration) taken from the core set indicator on urban air quality (CSI004).

3 European data bases on commuting

There are many sources of commuting data in the European countries which collect the data at country level, see the examples given in Chapter 1 and in Barrett et al. 2009. However, only few programs on the collection of commuting data are maintained by international organizations. Thus the collection of the primary data still takes place at the country level and the administration at international level provides a harmonized process of data collection and comparable results.
One of the international surveys offering commuting data is the Labour Force Study (LFS) maintained by Eurostat. LFS is a well established survey which is carried on in 26 European countries since 1983. Unfortunately the use of commuting data from LFS for a Europe-wide study has some disadvantages. LFS is in principle a selective survey not aiming for a complete coverage of the population. The aim of the LFS is to provide time-series with quarterly results. Another disadvantage of the LFS is in the coarse level of spatial resolution of the results. The data can be purchased for most of the countries at NUTS2 level (e.g. regions)only and for Germany, Great Britain and Austria even for the NUTS1 (e.g. provinces) only.

For this study commuting information has been obtained from the Urban Audit (UA). The following information was extracted[2] for each city (see Figure 3.1) from the database:

- Total residential population;
- People commuting into the city;
- People commuting out of the city.

Data was extracted for four time periods (1989-1993; 1994-1998; 1999-2002; 2003-2006). In case of missing population numbers, data was completed using the information collected under the Core Set Indicator CSI004 (Urban Air Quality).

The data showed that the ratio of *people commuting into the city* to the *total residential population* is fairly constant for each of the four time intervals. In case of missing values for the last period (2003-2006) the average value of the proceeding periods is taken. Within a country this ratio varies within a limited range with the exception of some capital cities. For example, for cities in Belgium, excluding Brussels, we found a ratio of 0.25 ± 0.05 but for Brussels a ratio of 0.50 is found. For cities in France, excluding Paris, the ratio is 0.12 ± 0.05 with a ratio of 0.06 for Paris. For cities for which none of the time interval commuting data was available a national averaged ratio for the other cities was inserted. For five countries (Greece, Latvia, Lithuania, Malta, and Romania) no commuting information at all was available. In these cases we used the same ratio as in a neighbouring country (Italy, Estonia, Estonia, Italy and Bulgaria, respectively). A similar approach was used for gap-filling missing data of the ratio of *people commuting out of the city* to *total residential population.*
As only limited data for the national level is available it is difficult to validate the gap filling method. Only for a number of Dutch cities (Table 1.1) data is available for comparison. For these cities a good correspondence has been found (R^2= 0.89), see Figure 3.2

The gap-filled database offers the possibility to estimate in a consistent way the day-time population in more than 350 of the larger cities. This fulfils partly the requirements of a model to estimate the exposure for commuters. Missing information is on the home address of the commuter, the travel distance and time and the transport mode. We are not aware of data sources where this information is available. In the next steps further assumptions have to be made (see Chapter 4).

[2] Data extracted on 21 May 2010. We acknowledge the assistance of Teodora Brandmuller and Kristina Dourmashkin in extracting the data.

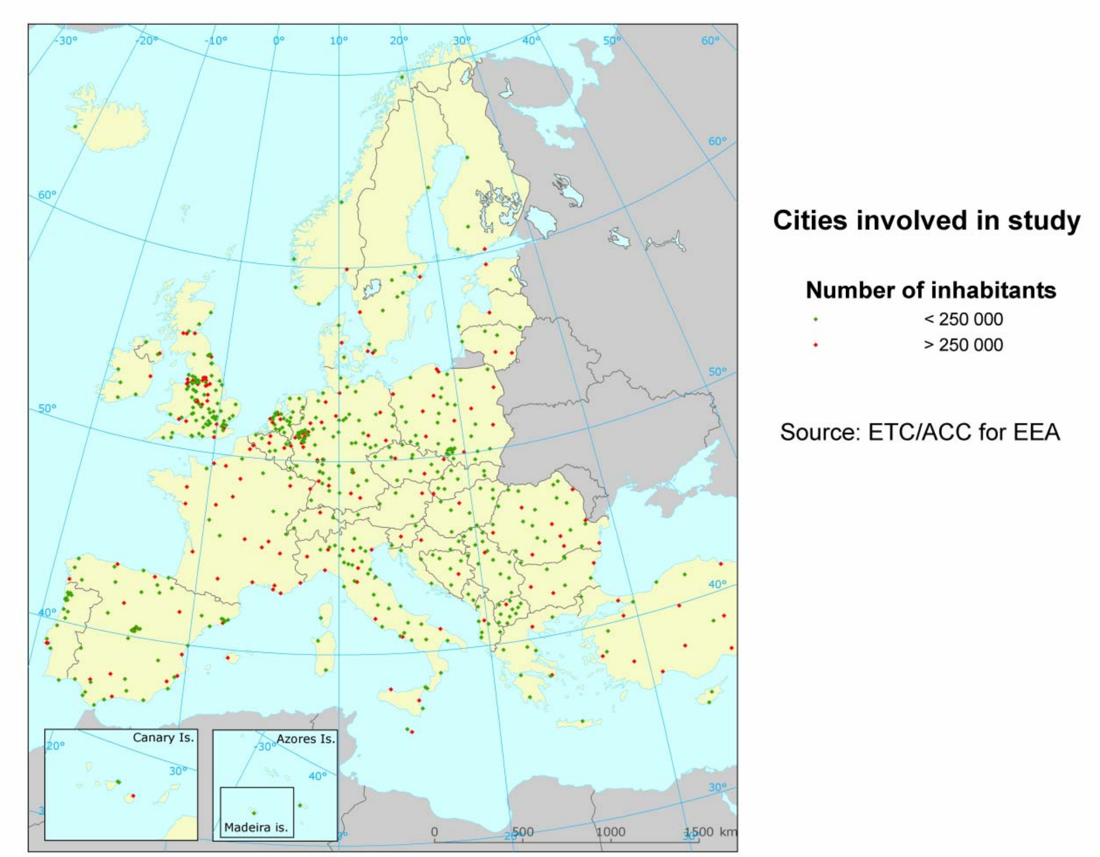

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Figure 3.1. Cities included in the Urban Audit dataset.

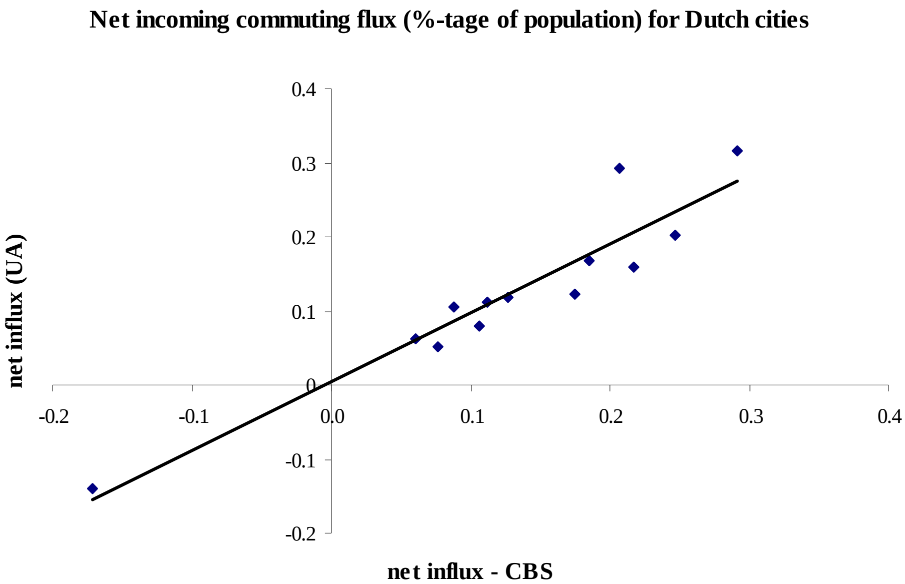

Figure 3.2. Comparison between the net incoming commuting flux (expressed as percentage of the urban population) as reported by the Dutch Statistical Bureau (CBS) (Vos and Trijssenaar 2000) and as extracted from the Urban Audit dataset.

4 European population exposure and the effects of commuting

The cases studied so far (London, Silesia, Oslo) did not provide a clear view on the effect of commuting on the population exposure. The detailed information on commuter-flows available for London might be difficult to extrapolate to other European cities. City size, national or regional characteristics may affect such a generalization. Therefore, we are looking initially for a more simple approach to get a generalized picture of the importance of commuting on population exposure. If this first analysis shows that commuting has a significant influence on exposure and health impact estimates, more refined approaches will be needed.
Compared to exposure estimates using a static population, estimates including commuting may increase the exposure in three ways: (i) increased exposure during travelling; (ii) higher concentrations at the working place compared to the home address and (iii) a further increase in concentration as daytime concentrations tends to be higher than night-time concentrations. A first approach to estimate the exposure is described below.

Two different situations are considered:
- A night-time situation: the population is assumed to be at home, the standard residential population distribution is used here. A country-wise scaling on the concentration is applied to account for the systematic differences between the annual mean concentration and the concentration averaged over the night-time hours (between 19 and 07 hour the next morning).
- A day-time situation: part of the population is assumed to commute from their living places in the more sub-urban/rural areas towards the city centre. During the day (07-19) this fraction is exposed to the concentrations in the city. Similar to the night-time map, a country-wise scaling for the systematic differences between daytime and annual mean concentration is applied. The fraction of the population commuting into the city is taken from the Urban Audit dataset (see Chapter 3). This map where the commuters are at their working place, results in an increased urban population and a reduced rural population.

Note that at this stage the exposure *during travelling* is not included.

The diurnal variation is estimated on the basis of the AirBase results. Data are used from all stations having hourly PM_{10} monitoring data for 2007; a data coverage of 75% for each hour of the day was required. A total of 918 stations fulfilled these criteria. To have a better match with human activities, the diurnal profiles are based on local time; a correction for summer time has been included. Day- and night time scaling are estimated per country and per station type (rural background, (sub)urban background).

Different PM_{10} diurnal profiles have been found, depending on station type and location, see Figure 4.1. Both at (sub)urban and at rural background stations the highest concentrations might be observed during the day or during the night. The relative diurnal amplitude is very similar for (sub)urban and rural background stations; day-time concentrations vary between 22% below the daily mean to 22% above the daily mean. The largest variation is found for the industrial stations where day-time concentrations are between 23% below to 36% above the daily mean. A further interpretation of these observed differences falls outside the scope of this paper.

The observed day-time and night-time concentrations are interpolated over Europe using the routine methodology developed by the ETC/ACM. The procedure used for interpolation of day-time and night-time PM_{10} concentrations follows the methodology recommended for PM_{10} mapping (de Smet et al., 2010). The method is based on applying a linear regression model followed by ordinary kriging of its residuals (residual kriging). Interpolated maps for rural and urban areas are created separately. The rural map is based on the rural background stations and the urban map on the urban and suburban background stations. For rural areas, the supplementary data used is EMEP model output (PM_{10} annual average), altitude, wind speed and solar radiation, and for urban areas only results

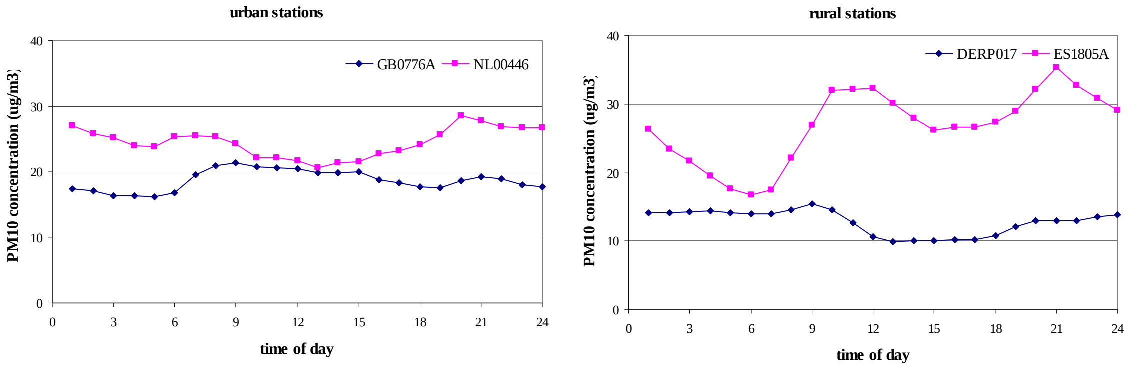

Figure 4.1. Examples of the PM_{10} diurnal profiles. Left at two urban stations in United Kingdom and the Netherlands, right at two rural stations in Germany and Spain. Data for 2007, extracted from AirBase.

from the EMEP model (PM_{10} annual average) are used. The spatial interpolation of residuals is carried out for each of those maps using ordinary kriging. Both the rural and urban maps are created in 1x1 km resolution. The merging of the rural and urban map into one combined final map is done on basis of an aggregated 1x1 km grid resolution of the originally 100x100 m population density grid (for detailed description of the method see de Smet et al. (2009) and references citied therein).

The resulting maps for 2007 are given in Figure 4.2. Two different levels of the colour scheme are used. Countries with more contrast colours have hourly values in AirBase so the diurnal variation could be calculated for them. The interpolation field for the other countries, symbolized with less

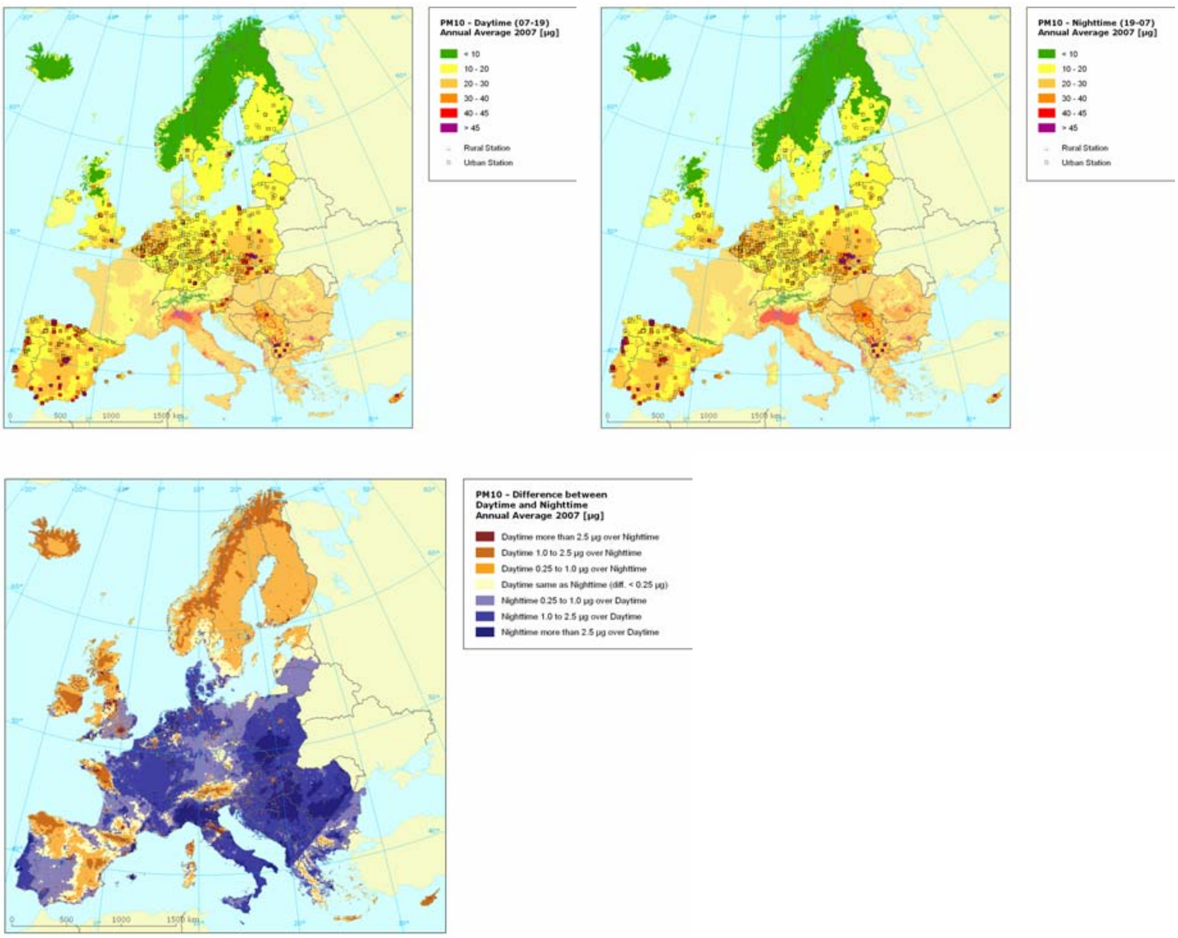

Figure 4.2. Interpolated maps of day-time (upper left), night-time (upper right) PM_{10} concentrations. The map at the bottom shows the difference between day-time and night-time concentrations.

contrast colours, is formed by the supplementary data and the interpolated values from the stations in the surrounding countries only.

Figure 4.2 shows, probably surprisingly, that the night-time concentrations are higher than the day-time concentrations for most rural areas in central and southern Europe, and in some urban areas as well.

The next step would be to generate population maps representative for the day- and night-time situations taking into account the change in the whereabouts of the commuter. Incomplete information precludes the construction of these maps. In Chapter 3 we described a commuter data base but information on the location of the working places and on the travel distance is not (yet) available.

However, to test, as a first approximation, the importance of commuting we develop a simple model (Figure 4.3).

We assume a region with a total population P_{tot} of which a fraction f_{urb} lives in a central city. Daily a net flux of people commuting into the city is given by F_{com} where F_{com} is expressed as a fraction of the resident urban population P_{urb} (where $P_{urb} = f_{urb} \cdot P_{tot}$) The number of commuters from the rural surrounding towards the central city is therefore given by $F_{com} \cdot f_{urb} \cdot P_{tot}$ with the restriction that this is less than the rural population ($P_{rural} = (1-f_{urb}) \cdot P_{tot}$)

We further assume a concentration ratio R of urban and rural concentrations. Finally, factors giving the increase in day-time concentrations for urban and rural situations are defined, d_{urb} and d_{rural}, respectively. The change in population weighted exposure, PWE, taking commuting into account can now be estimated.

The reference situation assuming a static population is given by:

$$PWE_{ref} = (P_{rural} \cdot C_{rural} + P_{urb} \cdot C_{urb}) / P_{tot}$$

When commuting is considered, the day-time PWE is given by:

$$PWE_{day} = \left\{ P_{rural}^{day} \cdot (1 + d_{rural}) \cdot C_{rural} + P_{urb}^{day} \cdot (1 + d_{urb}) \cdot C_{urb} \right\} / P_{tot}$$

where $P_{urb}^{day} = (1 + F_{com}) \cdot f_{urb} \cdot P_{tot}$ and $P_{rural}^{day} = \cdot P_{tot} - P_{urb}^{day}$

The night-time PWE is similarly calculated as:

$$PWE_{night} = (P_{rural} \cdot (1 - d_{rural}) \cdot C_{rural} + P_{urb} \cdot (1 - d_{urb}) \cdot C_{urb}) / P_{tot}$$

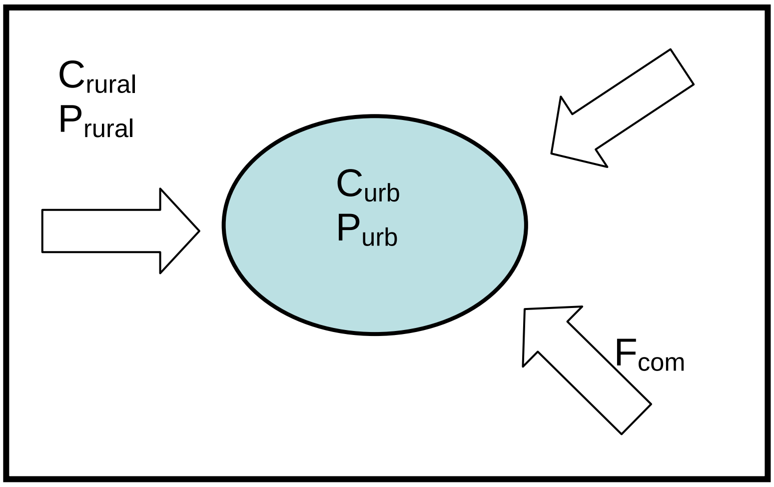

Figure 4.3.Schematic model to estimate the change in exposure when the effect of commuting is included.

The PWE including the effect of commuting is given as the average of PWE_{day} and $PWE_{night.}$

The highest effect of commuting on the PWE is to be expected when using an upper limit of the commuting fraction, F_{com} = 0.25 (see Chapter 3), assuming a densely populated city, f_{urb} = 0.75, a large difference in rural and urban concentrations (R = 4 , for example an urban concentration of 40 µg/m^3 and rural concentration of 10 µg/m^3) and assuming that the urban concentration is maximal during the day and that the rural concentration reaches its maximum at night (analysing AirBase data gives upper limits of d_{urb} = +0.22 and d_{rural} = -0.22). This results in an upper limit of the increase in PWE of about 12%; using a set of less extreme input parameters, increases around 5% are estimated. This is in line with the results from the case studies discussed above, where 4 case studies gave the increased PWE due to the commuting (changing daytime location) to be 0-12%.

However, for the commuters their individual exposure may strongly increase. On his home address the exposure equals C_{rural}. Accounting for the change in concentrations while working in the city, the exposure of an individual commuter is $0.5 \cdot (\cdot(1 + d_{rural}) \cdot C_{rural} + \cdot (1 + d_{urb}) \cdot C_{urb})$ in which we assumed that the person is 12 hours in the city. Using the worst case parameters the daily commuting routine may result in a tripling of the exposure of the commuter group. For more realistic situations increases by up to 50% can be expected when the exposure during travel is included as well. This is in line with estimates in some of the reviewed literature, as well as with results from the Oslo case study, see Chapter 1.

As discussed in Chapter 2 it might be that the commuters travel to a part of the city where concentrations are substantially higher than elsewhere in the city. We tested this in our simple model by adding 2 µg/m^3 to the concentration to which the commuters are exposed during working hours. This results in a small further increase in PWE. In the worst case scenario sketched above the increase is about 12.5% (reference case 11.8%), for the more realistic scenario we now estimate an increase 5.1 % (reference case 4.6%).

There is not enough data available to develop an estimate of a typical increased exposure of commuters as a result of the commuting travel itself. Further use of the London commuter data base could provide a useful example of such an increased exposure due to daily commuting travel for a large European city. If we assume that we spend per day 5% of the time travelling (EC 2004) and that the concentration during travel is 50% higher than the urban background concentration, then a further increase in PWE with a maximum of about 8% is estimated. However, one of the results of the EXPOLIS study is that, unexpectedly, time spent in different means of private (car, motorcycle) and public (bus, tram, metro, train) urban transport showed no consistent impact on $PM_{2.5}$ exposure in any of the cities included in the EXPOLIS project. How important the contribution from travelling is to the total exposure depends strongly on the transport mode and the pollutant (see e.g. Rim et al. 2008; McNabola et al. 2008)

5 Conclusions

The study on intra-urban variability of PM_{10} concentrations in European cities, based upon monitoring data in AirBase (Chapter 2), indicates that urban background concentrations show a variability of about 15% (95% confidence level) around the urban averaged concentration. The data suggest that within a city the local concentrations are not correlated with the resident local population densities. The previous detailed case studies summarised in Chapter 1 showed that improved spatial resolution of the urban concentration field, from 10x10 to 1x1 km, gave a modest increase in the population weighted exposure (PWE), of about 3% (for the Moravia-Silesia region and London). The results from these two approaches combined suggest that the intra-urban variability will have only a small/moderate increasing effect on the population weighted exposure, and thus on the estimated health impacts on the population. However, the city averaged concentration will lead to an underestimation of the exposure experienced by the fraction of the population at the high end of the exposure distribution.

At the population level the effect of commuting (shifting location from home to work/school) on the average population exposure (PWE), using a simplified approach due to lack of detailed data on commuting in European cities, was estimated to be less than 12% (Chapter 4). The case studies gave results of the same order, from 0 to up to 12% for 4 different cities (Moravia-Silesia, London, Athens, Oslo, in increasing order, Chapter 1). For an individual commuter the personal exposure may increase substantially, by 50% and more, based upon the simplified methodology. The Oslo case study similarly gave a 52% increased PM exposure as an average for 19 'test persons', with a maximum of 160% for one of the test 'persons', when the increased exposure both due to the shifting location and during the travel was considered.

Using the present WHO linear dose-response relationship for long term PM_{10} exposure, where the incremental health effect is independent upon the actual PM_{10} level, it is sufficient to take account of the commuting in terms of its effect on the average exposure when considering the impact of air pollution on the total population. The present study indicates that the effect of commuting on the average exposure in European cities should be taken into account; it could be of the order of 5%, varying from 0 and up to 10% or more for individual cities.

Thus, the fraction of the population exposed to the highest concentrations, the high end of the exposure distribution, is increased substantially by the commuters, their number is large and their increased exposure substantial. If it should be important for health effects reduction to limit the high end of the exposure, it is important to further investigate the size of the high-end population, their actual exposure, and to include commuting and commuters in exposure assessments.

The present study did not cover all topics that might be relevant for estimating the high end of the exposure distribution. The major shortcomings are:

- The real hot-spot situations (such as exposure in high traffic areas) are not yet accounted for in estimating the upper end of the exposure distribution.
- The effect of an increased exposure during commuting journeys is not yet included quantitatively. Information on travel time and transport mode is not widely available at the European scale.

Better estimates of the population exposure and its distribution would be helpful, as part of health effect studies, to improve the dose-response relations. It should to be noted, however, that the application of the better (more precise) information on population exposure would be limited at the present time if the risk coefficient from the studies listed in Table 1.3 would be used. All those studies have been based on the air pollution level at the place of residence of the study subjects as their exposure estimate. The health impact assessment should reproduce the conditions of the epidemiological study in order to avoid potential additional errors.

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