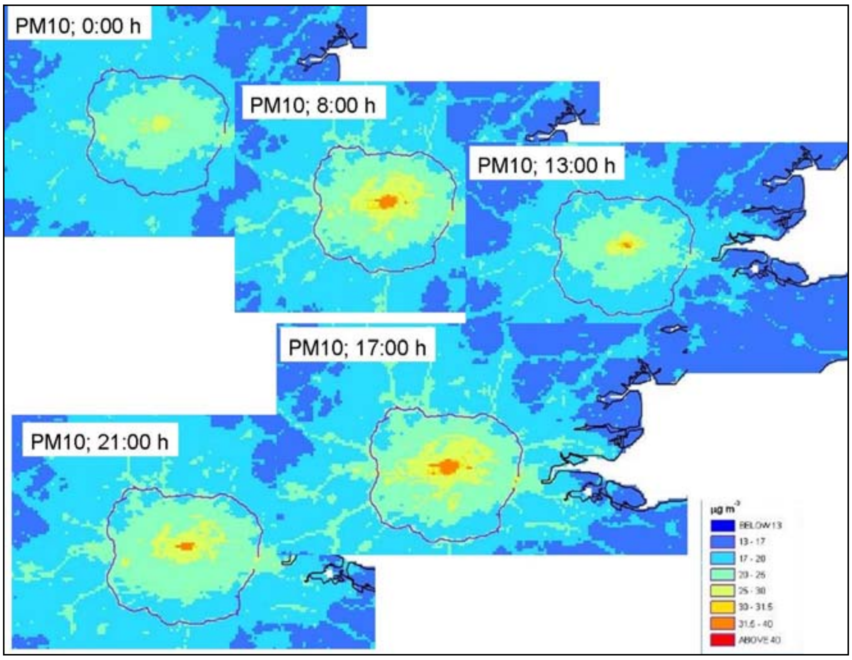
Air quality and health for urban influenced populations

Commuting and spatial scale as influences on estimated exposure/health impact



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European Topic Centre on Air and Climate Change

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Front page picture:
Hourly air quality maps for London (PM10). Source: case study in London published in this report.

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Abbreviations

AIRNET	Thematic network on air pollution and Health
APEI50	Accumulated Population Exposure Index (>50 g.m^{-3})
APHEIS	Monitoring the effects of Air pollution on Health in Europe
CAFE	Clean Air for Europe Programme
CAIR4HEALTH	Clean Air for Health
CBA	cost benefit analysis
CHAD	Consolidated human activity database
CLEAR	Cluster of European Air Quality Research
COMEAP	committee on the medical effects of air pollution
C-R	concentration-response
DEFRA	Department of the Environment, Food, Regions and rural Affairs
EC	European Commission
ENHIS	European Environment and Health Information System
ETC/ACC	European Topic Centre for Air and Climate Change
EU	European Union
ExpoFacts	European Exposure Factors Sourcebook
GAA	Greater Athens Area
HENVINET	Health and Environment Network
HIA	Health Impact Assessment
IDBR	Inter-departmental business register
LFS	Labour Force Study
LUR	Land Use Regression
MLSOA	middle layer super output area
OA	output area
OECD	Organisation for Economic Cooperation and Development
NPD	number of premature deaths
PCM	Pollution climate mapping models
$PM_{2.5}$	Particulate matter < 2.5 m diameter
PM_{10}	Particulate matter < 10 m diameter
SHAPE	simulation of human activity and pollutant exposure model
SOMO35	sum of means over 35ppb
STEMS	Space time exposure modelling system
TEOM	
TOTEM	time activity based exposure model
TMA	time microenvironment activity
TWA	Time weighted average
WHO	World Health Organisation

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1 Introduction

Assessments of the impacts of population exposure to air pollution upon the health of European citizens are currently performed by the European Topic Centre on Air and Climate Change (ETC/ACC) 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 underlying theme is to assess the potential importance of actual exposure patterns as compared to the current 10km approximation. Even if the ‘dose’ in dose-response relationships is to a large extent based on proxies for exposure, like stationary monitored levels, the space-time details of individual exposures could be utilised for more effective exposure-reduction measures.

Work on this topic in 2008 identified commuting as a factor which may modify the static exposure picture. In addition, significant temporal and spatial variations in the urban concentration field (e.g. Wilson *et al.*, 2005; Marshall *et al.*, 2008; as well as general experience) are overlooked at the current scale of assessment. Together, the identified factors can mean a notable departure from the estimated static exposure to the pollutant of concern.

Commuting implies two central departures from static residential exposure: a) time at work/school, and b) exposure during the commute itself. The relevance of these is determined by period and magnitude of exposure. It has been estimated that people spend, on average, some 30-40% of their time away from home (e.g., work, school), with 4-5% of this time in transit between different microenvironments, etc. There is an argument that during this time, they are often exposed to higher concentrations than they would be if they were located at their home address, and that the general pattern of commuting is from less polluted, more rural, outer areas towards more built up, industrial/commercial and densely trafficked inner areas of the city. This suggests exposure is greater than would be represented by the residential population distribution.

The case study for Oslo in 2008 suggested notable additional exposure for individuals who occupy high concentration locations during the day and who travel along high concentration routes, whilst case studies of London and Athens indicated the magnitude of air concentration variations which may exist in microenvironments, and the effect of assessing exposure based on a better spatial resolution (Barrett et al., 2008). The work in 2008 also suggested that commuting means exposure to higher concentration brackets for a limited percentage of the population, rather than more extensive exposure to existing brackets. Associated health effects would then be focused on an identifiable target group. Even if this group is proportionally limited, across Europe this may translate into large numbers.

At present, the baseline information on the scale of impact arising from commuting is simply missing and thus the size of the affected population and the scale of their additional exposure are unclear. The intention of this report is to build towards this robust estimate of the impact of commuting at European level, thus assisting in improving the macro and microscale basis for the health impact of air quality in Europe.

Intra-urban population movements, in particular, challenge the suitability of the typically 10km resolution assessments, noting that air quality may change considerably over 10km. The significance of a smaller scale spatial assessment was investigated in the 2008 case study of Moravia-Silesia, providing evidence that reducing spatial scale down to 1km resolution will notably increase estimates of exposure (Barrett et al., 2008). This work in 2009 also continues to explore the additional information available from smaller scale assessment.

From the planned objectives of the task "Air Quality and Health for urban influenced population", the study has addressed the following themes:

- Critical review and evaluation of available methodological approaches and their feasibility for European health impact assessment (HIA) addressing the effects of air pollution on urban-influenced population.
- Methodological and health impact analysis building on the CAFE Programme findings and the ETC/ACC 2008 study.
- Concentration on exposure impact assessment.
- A pre-study in respect of the health effects of particulates that can be used to support the EU Commission in the planned review/revision of Air Quality Directives in 2011/2012.

In addressing these themes, this study has in particular:

- Explored further the influence of commuting patterns identified in previous studies (Oslo; Barrett et al., 2008) on exposure in a large European city (Greater London Area); sought parallel/supporting evidence from other dissimilar urban areas; e.g. Moravia-Silesia in central-eastern Europe and Athens as a densely populated urban area in southern Europe.
- Identified difficulties that need to be addressed in undertaking such a health impact assessment at a European scale (population and mobility data).
- Reviewed the influence of assessment that different spatial scales may have on calculated exposure that was identified in the previous study.
- Identified useful future work to help construct a generalised framework for city-wide health exposure assessments across Europe.

The study has focused on PM_{10}. This pollutant is the one that likely has greatest impact upon urban influenced populations, and is the one for which most work on exposure assessment and health impacts has been conducted. Thus it supplies both the most relevant and the most reliable basis upon which the impact of commuting might be considered. Nevertheless, it is important to remember that the appropriate method and the conclusions of any exposure assessment are both highly dependent on the particular pollutant of concern. Ozone exposure, for example, would be anticipated to paint a dissimilar picture.

The findings of the 2008 study on the urban health impacts of air pollution are an important starting point. The conclusions of that study were:

- Accounting for smaller spatial scale exposure of an urban population will increase the estimated exposure of the total population above that estimated on the basis of a total urban population exposed to a single averaged air concentration. For ozone reversed situation (urban lower than rural) may occur.
- In addition, the movement of a city population daily towards the centre, including commuting on traffic corridors, increases the general exposure level. For a certain percentage of the population this increase will mean exposure to higher concentration brackets.
- Whilst the proportion of the population freshly exposed to higher concentration brackets may be small, it is expected that this translates into large absolute numbers for Europe as a whole.
- This smaller scale exposure still represents an underestimate of the proportion of the total European population subjected to higher exposure, as a significant fraction of the non-urban population commutes into urban areas daily. This group may be expected to be subject to substantial increases in exposure.
- Whilst daily movements may mean that the average exposure of European urban populations may be of the order of 20% greater than estimated by a simple urban average, such averaging may give misleading results. This increased exposure, and the potential health effect, will in reality be focused on an identifiable target group with a significantly increased exposure, rather than being averaged across a larger number.
- Regions of Europe experience different relationships between $PM_{2.5}$ and PM_{10} concentrations at rural, urban and trafficked locations, encouraging regionally specific assessments.
- When the exposure of the urban population in Europe to $PM_{2.5}$ is estimated, it is found that for 10 Member States the Average Exposure Index lies above the binding value for 2015 of 20 $\mu g/m^3$, and in 5 Member States it lies at or below this level. Whilst for 12 Member States the average exposure index is clearly below.
- Estimating the total number of premature deaths from exposure to particulate matter results in similar result when exposure is based on PM_{10} or $PM_{2.5}$ concentrations.
- Review of recent literature on the health impacts o exposure to $PM_{2.5}$ does not support updating the previous coefficient for estimating mortality of 6% per $10\mu g/m^3$ of $PM_{2.5}$
- There is evidence to suggest combined adverse effects of exposure to $PM_{2.5}$ along with exposure to ozone.

2 Review of Exposure Assessment Methods

This section provides an overview of the major recent developments in methodologies and tools for assessing exposure estimates in European urban areas. The focus is on exposure assessments, specifically the different methods for exposure estimates, their advantages and disadvantages for health studies, new approaches proposed in the last decade for reducing uncertainties and the emphasis on those areas in where research is needed.

2.1 Purpose of this review

Over the last decade, urban exposure assessment methods and the establishment of more accurate approaches have received increased attention. New findings have heightened awareness of the importance of both exposure and concentration, and the latest European directive on air quality (2008/50/EC) introduces new standards for $PM_{2.5}$ and mentions for the first time population exposure to air pollution as part of an air quality target (WHO, 2005).

Exposure assessments and air pollution impact assessments upon the health of European citizens are currently performed by the European Topic Centre on Air and Climate Change (ETC/ACC) based on 1) 10 km x 10 km grid cell averaged estimated air pollution concentration and 2) population represented by static residential address. Sufficient for approximate assessments, but it may be too inaccurate for detailed assessments. Firstly, there are significant temporal and spatial variations in the urban concentration field (e.g. Wilson *et al.*, 2005; Marshall *et al.*, 2008) which can be overlooked at the current scale of assessment. Secondly, human exposure depends not only on the air concentration of the pollutant but also on human factors, such as activity and behaviour which affect contact with the pollutant of concern. For instance, it has been estimated that people spend some 20-30% of their time away from home (e.g., work, school); and 4-5% of their time in transit between different microenvironment. Realistic exposure estimates of population in urban areas is a challenging task due the number of influencing factors which significance for the exposure assessment must be understood.

The objective of exposure estimation methodologies is to enable, in a practicable manner, an estimate that is as representative as possible of the real exposure situation. 'Practicable' implies, amongst other things, results that can be used for health impact assessments.

One of the issues raised by this latter need is the basis of the dose-response relationships used for health impact assessments: How is the dose defined? Is it based on monitored levels at a restricted number of stationary locations being used as proxy for the real exposure, or is it based on a more representative measure of the real exposure, e.g. based on personal monitoring or modelled space-time variations attempting to 'follow' individuals or population groups?

The main topic of this report is to look at methods that attempt to assess the real exposure situation, and to determine the significance of space-time departures from an average. Ultimately, the intention is to establish the benefit and opportunity for more targeted and more effective exposure-reduction measures.

Some of the factors influencing the estimates of air pollution impact have been previously identified (Barrett *et al.*, 2008). However, the establishment of new methodologies and approaches for exposure estimates constitutes one of the challenges of exposure science. The work synthesized in this section constitutes an overview of the state-of-the-art in exposure science and reviews some of the most relevant published studies and new developed methodologies. The focus is on methods which take into account commuting or human activity, and spatial scale. The state-of-the-art on exposure science and approaches to exposure assessment are summarized, followed by an overview of the work performed in European research projects, a synthesis of recent results on improved urban exposure assessment concerning spatial scale resolution and human activity, and finally an evaluation of exposure assessment methods in relation to their use in health impact assessment.

2.2 Approaches to exposure assessment

2.2.1 Definitions

According to WHO/IPCS Glossary of key Exposure Assessment Terminology (WHO/IPCS, 2004), exposure is the contact between an agent and a target, with contact taking place at an exposure surface over a certain exposure time period. Exposure can be expressed in different ways. Exposure is commonly expressed in concentration-time units and the magnitude is represented by the area under the time-dependent curve of the exposure concentration as (Lioy, 1990):

$$E = \int_{t_0}^{t_1} C(t)\,dt, \qquad (2.1)$$

where E is the magnitude of exposure, C(t) is the exposure concentration as a function of time (*t*) and $t_1 - t_0$ is the exposure duration.

Time-weighted average is also commonly used in exposure assessments and is defined as the time-weighted average (TWA) concentration of an agent with which a person/population comes into contact in different activities and in different environments. It is expressed as (Ott *et al.*, 2006):

$$E_{(TWA)} = \frac{1}{T}\sum_{i=1}^{k} C_i\, t_i \quad , \qquad (2.2)$$

where $E_{(TWA)}$ represents the exposure of an individual of pollutant X over *k* microenvironments, C_i is the concentration in the i^{th} microenvironment, t_i the time spent in the microenvironment and T is the total time. We understand for microenvironment any appropriate identifiable and distinct set of spatial and temporal circumstances. Population exposure is therefore the sum of all individual exposures.

Exposure assessment is the process of estimating or measuring the magnitude, frequency and duration of exposure to an agent, along with the number and characteristics of the population exposed (WHO/IPCS, 2004). Ideally, it describes the sources, pathways, routes and the uncertainties in the assessment. Exposure assessment provides scientifically based information to policy makers as an integral component of risk assessment (Figure 2.1) along with dose-response assessment, and risk characterization.

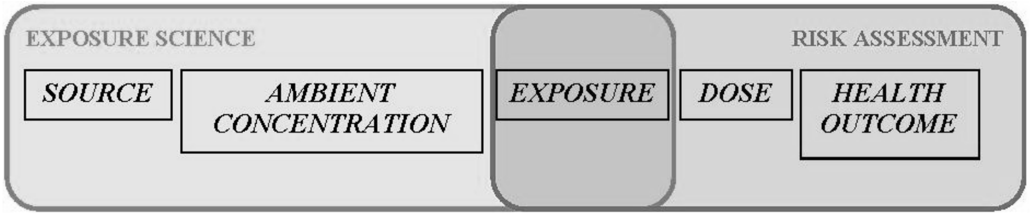


Figure 2.1: Scheme of the relationship between exposure science and risk assessment.

2.2.2 Approaches to quantification of exposure

Assessments of exposure to air pollution may vary depending on the type of study, pollutant and/or health impact to assess, and the quantification is therefore addressed in different ways. Exposure assessments are performed and quantified for individuals (*Individual exposure*) or populations (*Population exposure*). In the context of air quality directives and policy, the main concern is population exposure and in this particular report we will focus on *urban population exposure*. Exposure estimates are currently addressed by using different methodologies which may be roughly classified as *direct* or *indirect methods*.

Exposure monitoring are direct methods which vary over a broad range of methodologies, from personal exposure monitoring to biological monitoring. During personal monitoring programs different participants carry a device that registers pollutant level concentrations during the sampling time while the person moves between different microenvironments. In contrast, biological monitoring involves the use of biomarkers from which pollutant concentrations can be determined. Biomarkers are parameters taken from biological media such as tissue, fluids or cells and are used as indicator to provide evidence of exposure to and/or effects of one or more chemical pollutants.

Quantification of exposure by indirect methods is termed *exposure modelling* and is recommended for exposure assessment in epidemiological studies as opposed to personal monitoring (WHO and EC, 2002). Exposure modelling is generally classified into groups based on the type of mathematical or physical approach; see following section; "Classification of exposure assessment modelling" for more detail.

In order to unify principles, different guidelines of exposure assessment have been published such as the guideline from the US Environmental Protection Agency (US EPA, 1992) and the Principles of Characterizing and Applying Human Exposure Models (WHO/IPCS, 2005). However, as this report will show additional research is needed. Exposure assessments are currently performed by the Topic Centre based on 10 km x 10 km grid cell averaged estimated air pollution concentration and population data represented by static residential address. Concern about the accuracy and representativeness of this traditional method is well known and several studies have indicated the variability of urban environment and the factors affecting the accuracy of the method (e.g., Barrett *et al.*, 2008; Özkaynak, *et al.*, 2008; Beckx *et al.*, 2009). This report will focus on two of the factors of greater influence, namely spatial scale resolution, and population distribution and mobility.

2.2.3 Classification of exposure assessment models

Several studies have been published about exposure assessment modelling with emphasis on the review and evaluation of intraurban exposure methods (Moschandreas, *et al.*, 2002; Jerrett et al, 2005b; Zou *et al.*, 2009). Methods to assess exposure to air pollution vary from simple procedures to more complex and sophisticated operations. Based on the physical/chemical process approach models can be classified as mechanistic or empirical, and based on the mathematical approach they are classified as deterministic or stochastic. The classification of the exposure modelling is based on the combination of the two approaches as indicated in Table 2.1.

Table 2.1: The categories of exposure models (WHO/IPCS, 2005)

	Mechanistic	*Empirical*
Deterministic	Mathematical constructs of physical/chemical processes that predict an exposure for a set of inputs.	Statistical models based on measured input and output values.
Stochastic	Mathematical construct of physical/chemical processes that predict the probability density distribution of population exposure using full range of input data.	Regression-based models, where model variables and coefficients are probability distributions, which represent variability and / or uncertainty in the model inputs and parameters

Depending on the selected approach and on the input data and assumptions, exposure models can be classified as *proximity models*, *air dispersion models*, *land used regression models*, *hybrid models*, among others. They can fit various of the above categories, dependent upon the model construction. Evaluations based on proximity to a pollution source (*Proximity models*) represent the simplest method in intraurban exposure assessment. Proximity models are based on the assumption that exposure at location nearer to an emission source is higher than at a position further from the source. Different studies have associated proximity to pollutant sources with health effects such as respiratory diseases (e.g. Aylin *et al.*, 2001), lung cancer (e.g. Gauderman *et al.*, 2007), defects in offspring (e.g. Suarez *et al.*, 2007), stroke mortality (Maheswaran and Elliot, 2003). However, this method involves large uncertainties and high risk of exposure misclassification due to factors such as the assumption of isotropic dispersion of the pollutants or the absence of time-activity patterns (Jerrett *et al.*, 2005b). The transport of pollutants is an input to the *air dispersion models* (these usually fit into the above 'mechanistic' category), which incorporate in addition emission data and meteorological information in order to provide concentration of pollutants over space and time. Air dispersion models are common method for assessing human exposure in urban areas and are continuously being improved by combination with other tools such as geographic information systems (GIS), activity – based simulations, and inhalation or human intake models. The European Topic Centre on Air and Climate Change keeps an online catalogue and description of the atmospheric dispersion models developed in Europe (http://air-climate.eionet.europa.eu/databases/MDS/index_html).

Land use regression (LUR) models constitute one of the most extended methods for assessing exposure to air pollution. The model predicts pollutant concentration based on surrounding land use and traffic characteristics (e.g. Briggs *et al.*, 1997). The LUR model is a multivariate regression model in which the dependent variable is the monitored levels of the pollutant of

interest, and traffic, topography among other geographical variables are the independent variables (Gilliland *et al.*, 2005). One of the disadvantages of LUR is that it depends on a dense (and expensive) monitoring network and in addition air dispersion is not involved in the modelling approach.
Hybrid models, which consist in the combination of several models, such as dispersion and human inhalation models, may combine the strengths of individual methods and would make more effective exposure estimates and mitigate uncertainties (Zou *et al.*, 2009).
In order to take proper account of human mobility as part of the exposure assessment, new approaches have been suggested in the literature. Noteworthy are the models which take into account time-activity data, weighted time exposure in different microenvironment or dynamic models in which the combination of air quality models and commuting patterns provides a new approach for exposure assessment. These new methods will be discussed in section 2.4: "Improved urban exposure assessment: Recent Results".

2.2.4 Representativeness of exposure estimates

Different factors influence the accuracy and representativeness of exposure assessment estimates. Some of these factors are:

- Temporal trend variations;
- Subpopulation particularly susceptible to higher exposures (e.g. children);
- The accuracy and feasibility of necessary input parameters;
- Infiltration to indoors of outdoor generated pollutants;
- Indoor sources of pollutants of concern;
- Time-activity patterns of the population or subgroups;
- Spatial scale resolution;
- When assessing the exposure to pollutants such as PM, PAH, PCB, etc., the spatial and temporal variation in composition might be important.

The significance of some of these factors has been previously reported (WHO, 2005; AIRNET, 2005). Time activity patterns and spatial scale resolution are our main concern and therefore the focus of this report.
A preliminary exploration of factors influencing estimates of exposure and air pollution impact upon the health of European citizens was performed and synthesised in Barrett *et al.* (2008). The report had a first look into the effects of spatial scale resolution, the significance of micro-scale variability in pollutant concentration and the effects of population distributions and mobility as main factors. Based on different case studies, Barrett *et al.* (2008) established that accounting for smaller scale and the movement of the population toward the city centre increase the general exposure level.
Variability and uncertainty are naturally critical issues in human exposure modelling. We understand variability as the indication of the heterogeneity of an input parameter whereas

uncertainty reflects the lack of knowledge of the true value. Spatial mapping is nowadays an essential tool for exposure assessment. Itself it involves significant uncertainties associated with the monitoring techniques, modelling, interpolation methods, etc. (Denby *et al.*, 2009). Exposure estimates are commonly provided by combination of these concentration maps with population density maps. Thus uncertainties will also be associated with the population density map, due to the application to static populations, uncertainties resulting from time-activity patterns, human activity levels, and those in the concentration-response functions.

The time-activity patterns, i.e. population movements to/from various microenvironments, will influence exposure assessment estimates and introduce significant uncertainties. Shifting of the population to the city centre from the suburban and more rural surrounding areas is expected and, therefore, exposure of commuters will often be higher than that estimated from residential addresses. In addition, recent studies have indicated that air pollution variability within city might be larger than between cities (Jerret *et al.*, 2005a; Miller *et al.*, 2007). The determination of factors influencing exposure assessment estimates in urban areas is essential for the accuracy of risk assessments of urban influenced population.

2.3 Exposure assessment in European research projects

Over the last decade several European projects and programmes on air pollution and its effects on health have been supported and new insights gained into air pollution effects and new exposure assessment methodologies. Some of the research projects dealing with air pollution and its effects on health are listed in http://ec.europa.eu/research/environment/themes/projects_en.htm#2 and some of the most relevant are synthesised in Table 2.2.

Examples of projects concerning the estimation of exposure to air pollution in European cities include EXPOLIS (Jantunen, 1999), MACBETH (Cocheo *et al.*, 2000), PEOPLE (Pérez Ballesta, *et al.*, 2008) and URBAN EXPOSURE (Fløisand, 2006). Different methodologies were applied aiming to understand human exposure to air pollution. Personal exposure was considered by three of the projects. They considered exposure time durations of 12 hours (PEOPLE), 48 hours (EXPOLIS) and 108 hours (MACBETH), in combination with exposure modelling techniques or monitoring networks. The project URBAN EXPOSURE aimed to estimate personal exposure to particulate matter in an urban environment based on defined daily routes and micro-environment concentrations. In URBAN EXPOSURE, outdoor concentration was calculated using air dispersion model whereas the indoor concentrations were calculated on the basis of both outdoor concentrations and contributions from selected indoor sources. With regard to the pollutants considered, the EXPOLIS project focussed on a variety of VOCs, carbon monoxide and $PM_{2.5}$ and the importance of indoor emission sources, whereas MACBETH and PEOPLE focussed mainly on aromatic hydrocarbons. Some of the main achievements from these projects concern exposure assessment methods (Kruize *et al.*, 2003; Pérez Ballesta *et al.*, 2008) and the development of exposure assessment related databases, for instance the EXPOLIS database about human activity.

Some of the most relevant results obtained in these European projects in relation to improvements of urban exposure assessment methods are presented in Chapter 2.4; “Improved urban exposure assessments: Recent results”.

Significant amongst the European accomplishments has also been the establishment of networks and programmes on air pollution and health. The Clean Air for Europe (CAFE) programme of

the European Commission was established in 2001 with the aim to improve European air quality and reduce the health and environment impact due to air pollution. This programme has been central in instigating research into health effects of air pollution, in cooperation with WHO. In addition, *AIRNET* (A thematic network on air pollution and Health; 2002-2005), *ENHIS* (The European Environment and Health Information System), *APHEIS* (Monitoring the effects of Air pollution on Health in Europe), *ExpoFacts* (European Exposure Factors Sourcebook), *HENVINET* (Health and Environment Network), *CAIR4HEALTH* (Clean Air for Health – research needs for sustainable development policies) or *CLEAR* (Cluster of European Air Quality Research) are some of the most significant examples aiming to establish a long-term strategy dealing with air pollution and its effects on human health and the environment.
Some of the important advances and methodologies developed have resulted in the establishment of internet platforms or toolboxes for exposure and / or health impact assessment. One of the most relevant examples is the platform for exposure assessment supported by the Finish National Institute for Health and Welfare which compiles databases, modelling tools and information related to the field of exposure assessment (http://www.ktl.fi/expoplatform/home_ui/index.php). In addition, the toolboxes of the projects HEIMTSA or INTARESE will provide integrated methodologies and information (e.g. case studies, databases, general information and glossary) for environmental health policy following the full chain approach.

Table 2.2: Selected European projects addressing exposure to air pollution and/or the effects on health.

PROJECT	TITLE / WEB ADDRESS
AIR4EUR	Air4Eur
	http://www.air4eu.nl/index.html
AIRMEX	Indoor Air Monitoring and Exposure Assessment Study
APMoSPHERE	APMoSPHERE - Air Pollution Modelling for Support to Policy on Health and Environmental
	http://www.apmosphere.org/
ENVIRISK	Assessing the Risks of Environmental Stressors: Contribution to Development of Integrating
	http://envirisk.nilu.no/
ESCAPE	European Study of Cohorts for Air Pollution Effects
	http://www.escapeproject.eu/
EXPOLIS	European Exposure Assessment Project
	http://www.ktl.fi/expolis/
ExternE	Externalities of Energy
	http://www.externe.info/
FUMAPEX	Integrated Systems for Forecasting Urban Meteorology, Air Pollution and Population Exposure
	http://fumapex.dmi.dk/
HEIMTSA	Health and Environment Integrated Methodology and Toolbox for Scenario Assessment
	http://www.heimtsa.eu/
INTARESE	Integrated Assessment of Health Risk of Environmental Stressors in Europe
	http://www.intarese.org/
INTEGAIRE	Integrated Urban Governance and Air Quality Management in Europe
	http://www.integaire.org/
MacBeth	Monitoring of Atmospheric Concentration of Benzene in European Towns and Homes.
	http://www.integaire.org/database-new/examples/uploaded/view_example.php?id=546&c=&m
OSCAR	Optimised Expert System for Conducting Environmental Assessment of Urban Road Traffic.
	http://cordis.europa.eu/data/PROJ_FP5/ACTIONeqDndSESSIONeq112362005919ndDOCeq1399ndTBLeqEN_PROJ.ht
PEOPLE	Population exposure to Air Pollutants in Europe
	http://www.citidep.net/people/epeoplehome.html
SAPHIRE	Source Apportionment of Airborne Particulate Matter and Polycyclic Aromatic Hydrocarbons in
	http://www.ges.bham.ac.uk/sapphire/
SATURN	Studding Atmospheric Pollution in Urban Areas
	http://aix.meng.auth.gr/saturn/
SAVIAH	Small-Area Variations In Air quality and Health
URBAN-AEROSOL	Characterisation of Urban Air Quality - Indoor/Outdoor Particulate Matter Chemical
	http://tarantula.nilu.no/projects/urban-aerosol/
URBAN EXPOSURE	Urban Exposure
	http://tarantula.nilu.no/urban_exposure/

2.4 Improved urban exposure assessments: recent results

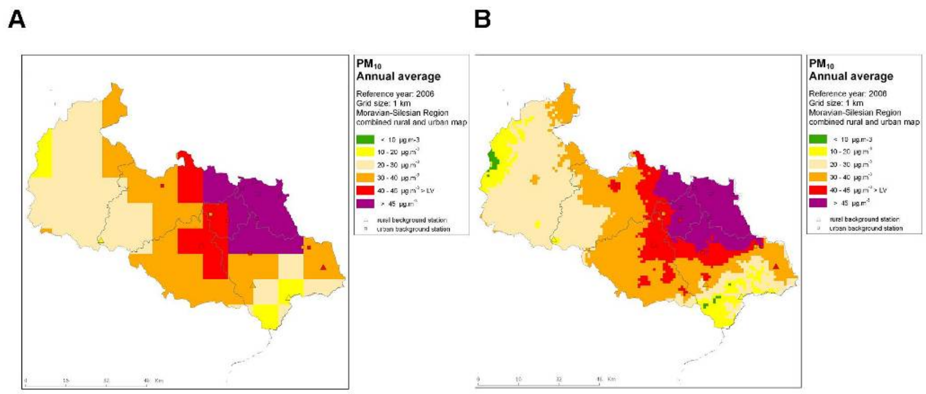
The increasing number of studies on methods and approaches for the assessment of human exposure to air pollution in European cities is a consequence of the growing interest between scientists and policy makers in exposure science, and the need to improve the basis for policy making regarding the improvement of air quality, especially in urban areas. This chapter will focus on advances published regarding the importance of human activity and spatial scale as factors influencing the accuracy of exposure assessments.

2.4.1 Spatial resolution in exposure modelling

Urban air pollution models typically provide calculated concentration estimates at a number of points in the modelling area, often called “receptor points”. Usually the model gives concentrations in a system of square grid cells, the concentration value valid for the centre or the average of the grid cell. Some models also give concentrations in otherwise specified points, these could be points near the streets or near other sources. The size of the grid cells is termed the spatial resolution of the model. When calculating exposure based upon the modelled concentrations, the population within each grid cell is normally given the concentration of the cell as their exposure, for the modelled time period. The smaller the grid cell, within a reasonable scale range, the better is the accuracy of exposure assessment of a static population, if indeed the resolution of the population matches that of the grid size. Thus, the spatial scale resolution used is important for the accuracy of the exposure assessment. A prerequisite for the validity of this statement is that the model accuracy does not decrease with smaller grid size. This is often the case, so the spatial scale question is a complex one.

The selection of spatial resolution and its importance for exposure and / or environmental impact assessment has been addressed in few studies (e.g. João, 2002; Stroh *et al.*, 2007; Barrett *et al.*, 2008). Intraurban air pollutant concentrations vary spatially and temporally (e.g. Wilson *et al.*, 2005; Marshall *et al.*, 2008) depending on factors such as wind direction or the geometrical characteristics of the main city features, among others. The development of new models for predicting the spatial intraurban variations of air pollution has been the focus of several studies (Brunekreef and Holgate, 2002; Moore *et al.*, 2007; Su *et al.*, 2008; Wheeler *et al.*, 2008). According to the scale and gradient of the variations, spatial resolution constitutes an important factor for reducing the associated uncertainties.

João (2002) showed how the significance of air pollution impact and the number of houses affected by roads varied according to the geographical scale. The variations based on scale choice can have important repercussions for the accuracy of exposure assessment and subsequently for the accuracy of health impact assessment studies. Barrett *et al.* (2008) reported the results obtained by the assessment of locations in Athens, central London, and Silesia aiming to evaluate the comparison between assessing the health impact of air pollution at city scale and at neighbourhood scale. As an example, Figure 2.2 shows the differences between PM10 annual average concentration fields estimated at 10 and 1 km grid size and the exposure population estimates (Figure 2.2C), for the Moravian-Silesian region. These authors concluded that accounting for more detailed geographical scale (1x1 km grid) will increase the estimated exposure of the total urban population on a 3.26% as compared with estimates based on a total urban population exposed to a single averaged air pollutant concentration (10x10 km grid).



C

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

Figure 2.2: PM10 annual (2006) average concentration fields at 10 km (A) and 1 km (B) grid size in Moravian-Silesian region and population exposure estimates (C) (Barrett et al., 2008).

The spatial scale at which intra-urban studies are undertaken may affect the outcome of the assessment, and underestimations by using coarse grid are expected. However, the performance of a study at a desired spatial scale will depend on factors such as data availability and / or budget.

2.4.2 Human activity on exposure modelling

Human exposure in urban areas depends not only on the concentration of the pollutant, but on human factors such as movements within the area, spend time in different microenvironments, activity levels and behaviour, which influences the concentration of the pollutant of concern that each individual meets and burdens his body with.

Time-activity patterns

Time spent in different urban microenvironments is particularly of interest, since concentrations may vary significantly, whether one is close to a street or another source, in the city centre or in a suburban residential area. Results from studies on how the population spends time in different microenvironments are given below.

Figure 2.3A shows the variation of the population activity in Helsinki between October 1996 and December 1997 (Kousa *et al.*, 2002) obtained as part of the European project EXPOLIS (Table 2.2). The population spends a significant fraction of time indoor, at home and in the work environment. Figure 2.3B shows the number of persons in different microenvironments in Lisbon city centre (Borrego *et al.*, 2006).

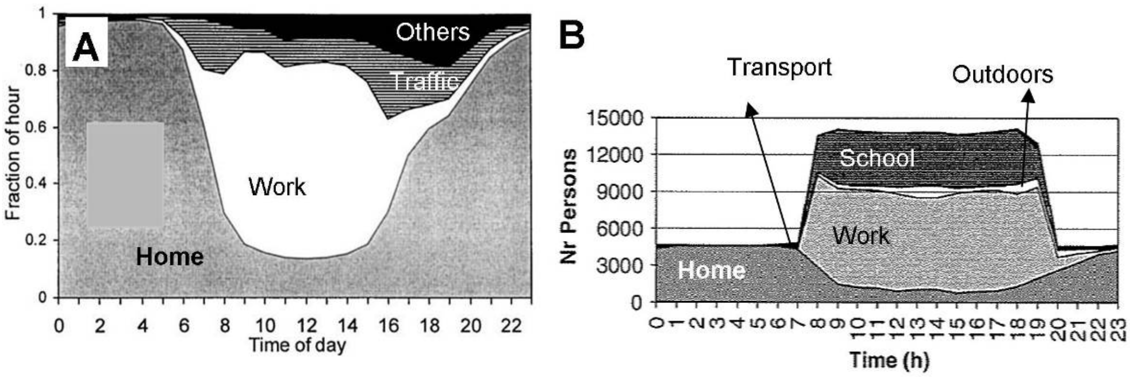


Figure 2.3; A: Diurnal variations of the activity of the population in Helsinki Metropolitan Area (October 1996 - December 1997; Kousa et al., 2002). B: Number of person per microenvironment within Lisbon (Borrego et al., 2006).

Studies on this are also available from the European Commission (EC) and OECD (2008). Table 2.3 shows results on time-activity patterns from the reporting to the EC from various countries (EC, 2004). Approximately 20-30% of the time is spent away from home (e.g. work, school) and 4-5% in transit between different microenvironments (Table 2.3). Time spend at home dominates the 24-hour time-activity pattern of European citizens, including evening and night hours. The OECD (2008) data are shown in Table 2.4, which supports the EC data: 4-5% of daily time is spent commuting. Time spent travelling or commuting to and from work is an important part of the daily activity pattern, taking place at times (rush hours) of typically high air pollution levels.
The average time spent on an activity (Table 2.3) is estimated from data reported as mean for whole group of persons across the whole year. The mean included all persons, whether they have performed the activity or not, all days of the week, working and holiday periods. “Indoor other” includes other people’s home, restaurants, cafe and pub. “Other” includes unspecified locations and weekend house. The commuting time from OECD (2008) and reported in Table 2.4 are averages estimated from adults who commute from and to work without child, with child under age and with school aged children.

Table 2.3: Average time spent on an activity (min day-1) (EC, 2004).

Location	year	Daily Travel	Home	Work / School	Indoor other	Other	Away from Home (%)
Germany	2001-2002	80	977	158	42	183	32
Belgium	1998-2000	82	1034	155	34	135	28
Estonia	1999-2000	65	1017	34	221	104	29
Finland	1999-2000	74	967	51	175	174	33
Hungary	1999-2000	56	1084	182	34	83	25
Norway	2000-2001	80	942	47	214	157	35
Slovenia	2000-2001	81	1044	27	199	90	27
Sweden	2000-2001	78	970	39	212	141	33

The moving of large population groups into urban centres during work hours is, in addition to the actual commuting and the increased exposure they get there, another main effect which influences the total population exposure compared to the exposure estimates at home address. Some studies have established an increase in the city centre occupants during work hours of two or three times the residential population due to commuting patterns (Borrego *et al.*, 2006; Barrett *et al.*, 2008). The US Census Bureau highlighted the variations of daytime population due to commuting in several cities (U.S. Census, 2000). Table 2.5 shows the percentage of increase due to commuting in some American cities based on the number of people present in an area during business hours and the residence population present during the evening and nighttimes. Amount of commuting generally increases with city population (Wash DC is a special case, situated within a heavily populated urban agglomeration).Table 2.5 shows in addition the population increase in Dutch cities based on the flux of workers to cities (Vos and Trijssenaar, 2000) and the residence population (Statistics Netherlands, 2001). Cities like Amsterdam and Rotterdam have commuting amount similar to large US cities. The Dutch data also shows that for very small cities, commuting can be very limited. The daily commuting between different areas constitutes an important factor of modern times and commuting between residence address and working place may constitute one of the main factors influencing exposure assessment estimates. Assuming that we spend per day 5% of the time travelling and 10% at the work place (see Table 2.3), and assuming that the concentration during travel and at the working place is 50% higher than at the residence, it is estimated that by taking the exposure at residence address the “real exposure” may be underestimated with at maximum 7.5%. Consequently, health impacts may also be underestimated.

Table2.4: Daily average and percentage of time spent travelling in different locations. M / W: Men / Woman.

hour day-1	%	Location	Reference	Notes
1,25	5,21	UK	National Statistics, 2003	Travelling time
1,20	5,00	Northampton, UK	Briggs et al., 2003	Adults
1,30	5,42	Northampton, UK	Briggs et al., 2003	College students
1,00	4,17	Northampton, UK	Briggs et al., 2003	Schoolchildren
1,70	7,08	Germany	Seifert et al., 2000	adults
1,40	5,83	Germany	Seifert et al., 2000	Children
1,30	5,42	USA	Klepeis et al., 2001	time "in vehicle"
1,23	5,13	Norway	Statistic Norway, 2008	2000
1,15 / 1,00	4,79 / 4,17	Belgium	OECD, 2008	M / W, 2006
1,07 / 0,95	4,44 / 3,96	Bulgaria	OECD, 2008	M / W, 2006
0,96 / 0,90	4,03 / 3,75	Estonia	OECD, 2008	M / W, 2000
0,78 / 0,77	3,26 / 3,19	Finland	OECD, 2008	M / W, 2000
1,03 / 0,93	4,31 / 3,89	France	OECD, 2008	M / W, 1999
1,10 / 0,85	4,58 / 3,54	Germany	OECD, 2008	M / W, 2000
1,07 / 0,95	4,44 / 3,96	Italy	OECD, 2008	M / W, 2003
1,37 / 1,13	5,69 / 4,72	Japan	OECD, 2008	M / W, 2006
1,18 / 1,03	4,93 / 4,31	Latvia	OECD, 2008	M / W, 2003
1,03 / 0,93	4,31 / 3,89	Lithuania	OECD, 2008	M / W, 2003
0,95 / 0,83	3,96 / 3,47	Norway	OECD, 2008	M / W, 2001
1,02 / 0,97	4,24 / 4,03	Poland	OECD, 2008	M / W, 2004
0,85 / 0,82	3,54 / 3,40	Slovenia	OECD, 2008	M / W, 2001
1,08 / 1,05	4,51 / 4,38	Spain	OECD, 2008	M / W, 2003
0,82 / 0,78	3,40 / 3,26	Sweden	OECD, 2008	M / W, 2001
1,07 / 0,85	4,44 / 3,54	UK	OECD, 2008	M / W, 2006
1,32 / 1,02	5,49 / 4,24	USA	OECD, 2008	M / W, 2006
1,05	4,38	Canada	OECD, 2008	Men, 2005
0,72	3,01	Australia	OECD, 2008	Men, 2006
1,04	4,33	*Average*		
0,20	0,82	*Standard Deviation*		
1,70	7,08	*Maximum*		
0,72	3,01	*Minimum*		

Table 2.5: 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

Examples of exposure models that incorporate time-activity patterns

Several studies emphasized the importance of population movements between different microenvironments but few studies analysed how to implement this in exposure modelling and finally in health impact assessment methodology. Pérez Ballesta *et al.* (2008) proposed, based on the results from the EU PEOPLE project, an application based on a regression model that use time-microenvironment-activity data (TMA) data from diaries expressed as:

$$EC_i = \exp\left[A_i + \sum_j \left(a_{i,j} t_j + b_{i,j} t_j^2\right)\right] \qquad (2.3)$$

where EC_i is the exposure concentration of the compound "*i*", t_j is the time spent in the activity "j", A_i, $a_{i,j}$ and $b_{i,j}$ are the regression coefficients corresponding to the compound *"i"* and the activity *"j"*. Exposure concentration (EC_i; Equation 2.3) can be considered as the product of two factors: a baseline exposure concentration level (BS_i) represented by $\exp[A_i]$; and a total activity factor, AT_i resulting from the contribution of all considered activities:

$$AT_i = \prod_j AT_{i,j} \qquad (2.4)$$

where

$$AT_{i,j} = \exp\left[a_{i,j} t_j + b_{i,j} t_j^2\right] \qquad (2.5)$$

According to the approach proposed by Pérez Ballesta *et al.* (2008), in the absence of information about activities, the baseline would represent the average exposure concentration.

Borrego *et al.*, (2006) developed a methodology to estimate population exposure areas based on the combination of information on concentrations in different microenvironments and population time-activity pattern data. The integrated exposure is expressed as an Accumulated Population Exposure Index (APEI) which estimates the number of persons exposed to a pollutant above a certain concentration level. The index was developed in an assessment of human exposure to PM_{10} and the daily limit value of 50 µg m^{-3} was considered, so the accumulated population exposure index is expressed as:

$$APEI50_i = \sum_t^{24}(C_{ti} - C_{threshold}) \times P_{ti} \qquad (2.6)$$

where $APEI50_i$ is the accumulated exposure index (µg m^{-3} person h) for the population exposed to concentration above 50 µg m^{-3}, during time *t* for grid cell *i* and *P* is the number of persons exposed. According to Borrego *et al.*, (2008), the accumulated index could be a good short-term and long-term exposure indicator. However, exposure assessment is an integrated component of the health impact assessment and the use of the accumulative index in dose – response or concentration – response functions need further studies.

One of the main goals of this report is to ascertain that commuting to and from work influences exposure assessment. Commuting clearly induce significant variations in city population between working hours and evening / nighttimes (Table 2.5). In addition, commuting take place in relatively polluted microenvironments such as within road vehicles or foot/cycle close to busy roads. Gulliver and Briggs (2005) developed a GIS-based dynamic exposure model to estimate journey-time exposures (STEMS – Space-Time Exposure Modelling System). The system estimates individual or group exposure to air pollution based on the combination of source activity (source activity / emission model; SATURN), pollutant dispersion (atmospheric dispersion model; ADMS-Urban) and travel behaviour (time-activity based exposure model; TOTEM). These authors compared the modelled exposure for 50 children based on concentration at the home residence, home plus school and the integrated exposure across journey, school and home microenvironments. The exposure estimates were rather similar and the highest exposure was obtained for the integrated exposure (i.e. home plus school and journey; PM10: 18.8 ± 1.5 µg/m^3), obtaining an approximate 8% increase with regards to the exposure estimate based on home address (PM10: 17.4 ± 1.2 µg/m^3).

One of the most critical aspects of exposure assessment and in particular of studies concerning human activity is data availability. Time-activity pattern data may be accessible from databases such as the National Human Activity Pattern Survey (NHAPS – Klepeis *et al.*, 2001) and the Consolidated Human Activity Database (CHAD – McCurdy *et al.*, 2000). However, both databases are from USA and they may not be representative of European conditions. Alternatively data may come from time-activity models such as the Simulation of Human activity and Pollutant Exposure model (SHAPE – Ott *et al.* 1988) or activity-based models such as ALBATROSS (A Learning-Based Transportation Oriented Simulation System) used to evaluate exposure to air pollution in a Dutch urban area (Beckx *et al.* (2009).

The absence of human activity in exposure modelling may underestimate exposure to air pollution in urban areas. However, the implementation of behavioural factors such as commuting between residential address and working place in exposure modelling is a challenging task which needs further research; recent data on activity patterns is not available at European level.

The effect on exposure of taking account of time-activity patterns

Several studies have addressed and evaluated exposure based on human activity and concentration levels (Clench-Aas *et al.*, 1999; Kousa *et al.*, 2002; Kruize *et al.*, 2003; Gulliver and Briggs, 2005; Borrego *et al.*, 2006; Pérez Ballesta *et al.*, 2008; Özkaynak *et al.* 2008; Beckx *et al.*, 2009, among others) and some of these studies have reported increases of 50 and 100% on exposure estimates with regards to those based on static residential address (Özkaynak *et al.* 2008; Pérez Ballesta *et al.* 2008; Barrett *et al.*, 2008). In exposure models used and developed in these studies, the time spent in each microenvironment by a subject is combined with the concentration in the microenvironment. However, the way in which human activity is taken into account and combined with the concentration field varies between studies. For instance, human activity in exposure estimates has been based on data from time-microenvironment-activity (TMA) diaries (Kousa *et al.*, 2002; Pérez Ballesta *et al.*, 2008), activity patterns (Clench-Aas *et al.*, 1999; Borrego *et al.*, 2006) or activity-based transport model (Beckx *et al.*, 2009). Other studies compare static home and work population, select individuals as examples, or evaluate the exposure based on static home populations along with a statistical description of the variation in order to account commuting factors (Barrett *et al.*, 2008).

Özkaynak *et al.* (2008) showed the importance of incorporating factors related to time-activity, commuting and/or exposure factors on the exposure estimates. These authors compared the results obtained by classic exposure assessment, based on outdoor concentration predictions, and those results obtained from an inhalation exposure model. The concentration ratio obtained by the inhalation model versus the air quality dispersion models was greater than 1 for most traffic-related pollutants (i.e. 1.14, 1.19 and 1.24 for acetaldehyde, benzene and formaldehyde, respectively). Based on these results they established that commuting to work locations could elevate personal exposure to many pollutants to levels above twice those modelled from home addresses. In addition, they emphasized the importance of applying exposure assessment methods which incorporate time-activity data and/or commuting. Similarly and based on the results from the EU PEOPLE project, Pérez Ballesta *et al.* (2008) emphasized the importance of considering activities and locations for understanding human exposure, and proposed an approach for the exposure assessment of urban population. In this study, population exposure to benzene for commuting groups was approximately 1.5 times the city background (±50% increase) and 0.6 times the maximum hot stop values. Furthermore, a study performed in Oslo (Norway; Barrett *et al.*, 2008) showed that there is more than twice people entering Oslo to work each day than leaving, indicating that the population is shifted towards the city during working hours. The exposure assessment was performed considering 19 hypothetical individuals as exposure scenarios and an increase of 100% in exposure concentration for commuter compared to the exposure concentration at home address was assumed as reasonable. However, the increase in average yearly exposure for these individuals would be approximately 20% compared to the average exposure based on residential address.

3 Air quality, health and commuting: Three case studies

Whilst the 2008 study clearly identified the potential for commuting to have an important influence upon estimated exposure, the principal case study at that time (Oslo) was of a relatively small city, with sharp urban-rural concentration gradients, and being at the head of a fjord has a very distinct physical geography which may have contributed to the apparent conclusions. Amongst the intentions for 2009 was to extend the study to large conurbations, to provide an examination of the exposure consequences of commuting across a range of European environments, and to explore the possibilities that exist with respect to data availability on a large scale. The following case studies were selected:

- Greater London, as a significant North-Western European city
- Moravia-Silesia, as a major population and industrial centre in Central–Eastern Europe
- Greater Athens, as a large and densely populated urban area in Southern Europe

In this section the methodological approaches taken in these case studies is set out, beginning with estimation of ambient air concentrations, running through the handling of population data, and concluding with the estimation of exposure itself and of health effects. The findings are presented in Chapter 3.2.

3.1 Methodology

3.1.1 Ambient Air Quality

Ambient air quality has been assessed variously on the basis of monitoring and modelling. Here an overview of the different approaches is given for intercomparison.

London: AEAT's Pollution Climate Mapping (PCM) models (Grice *et al.*, 2008) are used configured so as to produce annual average concentrations for each of the 24 hours in the day (i.e. 24 separate maps), using 2007 meteorological data. Road traffic emissions have a diurnal profile, whilst non-road traffic emissions are given a constant daily emission profile. Dispersion in the model accounts for a classification of ten land use types, and the model resolution is 1km x 1km.

Moravia-Silesia: Data taken from the automated ambient air monitoring network in the Czech Republic, with subsequent linear regression of measured and supplementary data (the mean annual average of official PM10 maps for the Czech Republic 2007 and 2008) and interpolation of regression's residuals is used to produce maps of average air quality in Moravia-Silesia for four daily time periods (night: 1800-0500; morning travel: 0500-0700; daytime: 0700-1600; evening travel: 1600-1800) for each quarter in the year (January-March; April-June, July-September, October-December) i.e. 16 maps in total. The linear relation of the measured and supplementary data had an R^2 value for each time period between 0.73 and 0.98.

Athens: The OFIS model from the European Zooming Model system was used configures to produce annual average concentrations for each of the 24 hours in the day (i.e. 24 separate maps), using 2000 meteorological data. Emissions are applied for all SNAP sectors. Advection is represented within two vertical layers – the first to 90m over the surface, and the second to the boundary layer. Model domain is 150km x 150km, and the output has a resolution of 1km x 1km.

3.1.2 Population distributions and commuting

London: The datasets used in this study were primarily derived from the workfiles of the 2001 Census of Population on place of usual residence and place of work (of their main job) for all people aged 16-74 (included students in employment), but also from some other sources, including the UK Inter Departmental Business Register (IDBR). Both the place of residence and place of work datasets were available at the highest spatial resolution (UK Census Output Area [OA] level – smallest unit with available data, typically 40-100 socially homogenous households) but this analysis predominantly used data aggregated to Middle Layer Super Output Areas (MLSOA – aggregations of OA's to similar size with a typical population of 7,200), from which unpublished origin-destination commuting data were analysed to provide information on the flows of different types of commuters between MLSOAs. This level of analysis allowed patterns of resident proportions and commuting to be explored locally, showing how they differ across Greater London.

A database stored and related commuting flows in, out, and within London and neighbouring regions. Simple statistical methods were used to estimate the commuting flows. Much of the data used to compile the database was taken from the special workplace statistics of the UK 2001 Census of Population and its related CommuterView datasets, providing detailed data and information on places of residence and place of work, thereby giving insight into commuting patterns.

The MLSOA commuting datasets were spatially analysed using GIS algorithms. For data output and spatial integration with the outputs (i.e., typical hourly 1km^2 background air quality maps for PM_{10}) from the Pollution Climate Mapping models, the spatially analysed datasets were aggregated at the 1km^2 grid level. Using GIS overlay analysis (i.e., spatial analysis by overlaying the MLSOAs' commuting data with the 1,604 1km^2 grid cells in Greater London), each 1km^2 grid cell was allocated a value for the estimated residential and daytime population - where a MLSOA covered more than one 1km^2 grid cell, the value for the estimated residential and daytime population was divided between the relevant 1km^2 grid cells based on the proportion of the area covered by the pertinent MLSOA. The spatial distribution of resident and daytime populations at the MLSOA and at the 1km^2 level in Greater London is given in the Appendix.
For the purpose of this study London has been subdivided into three regions - central, inner and outer London. These regions follow administrative district boundaries. The boundaries of these sub-regions are shown in Figure 4.1, and the associated administrative districts listed in the Appendix.

Table 3.1: Summary population statistics for Greater London

Population group	Population
A. Population (all ages)	7,172,091
B. Lives in London and works outside London	236,018
C. Lives and works in London	3,083,116
D. Lives outside London and works inside London	722,539
Net flow (**D**-**B**)	486,521
Workplace population (**C**+**D**)	3,805,655

Moravia-Silesia: The population distribution for the Moravia-Silesia region (see Appendix) is based on data collected by the Czech Statistical Office for the population registry of January 1st, 2009. The base data is available at the statistical unit LAU2, but as all maps were constructed at 1x1 km scale the population data was converted to the 1x1 km grid using the relative proportions of built-up land in the grid cells. In terms of commuting, the population distribution map may be understood as the 'night-time' population map without respect to daily commuting.

A map of 'commuting balance' was constructed using the population map and commuting data. At first, the number of people commuting daily to their work or school outside each municipality on the LAU2 level is subtracted from the population, and the numbers of people commuting daily to workplaces and schools inside each municipality is added.

Athens: The population data used was based on the 2001 National Census study obtained from the National Statistical Service. The population data is at a scale of 1 km × 1 km, with more specific information for the resident population per building block. This data was manipulated within a GIS package to attribute population data to each 1km grid of the air quality modelling mesh. In effect the value for the population in each cell is the sum of the populations of each building block contained in the cell. The population distribution over the Greater Athens Area (GAA) is presented in the Appendix. Grey scale areas represent areas with no inhabitants. The GAA has a population of around 4 million, and is the area with largest population density and economic activity in Greece. Athens is monocentric, with 25% of the population living, and 35% working in the centre. Working places are concentrated on radial arteries leading to the centre. Particularly high urban residential densities partly explain the lack of suburbanisation. Traffic congestion is severe and traffic speeds low.

In order to estimate the population commuting patterns on daily population densities, population flow data were used. The population flow information data are based on the difference between the number of people that enter ("inflow") a specific municipality and the number of people that leave from this municipality. As a result it is possible to calculate the net value of the daytime population in each area within the boundaries of the GAA, which is defined as the net residential population with addition of the inflow population and the subtraction of the outflow population. It should be noted at this point however, that although information for the static population is provided on a building block basis, information for the population flows is estimated on community – municipality basis. As a result, in cases when these areas are larger than 1 km^2 (or in general larger than the area of the grid cells of the domain), in order to allocate a value of the net daytime population an area weighted calculation was performed for the accurate distribution of the population flow in the cells that comprise the aforementioned area. Alternatives are available, e.g. Jobs-employment balance (JEB) Indicator which defines the ratio of jobs in a municipality to the number of municipal inhabitants employed within the study area (Milakis *et al.*, 2008; Cervero, 1988; Frank and Pivo, 1994; Messenger and Ewing, 1996; Kockelman, 1997).

3.1.3 Exposure and health impact methodology

Estimating Exposure

Exposure can be calculated via a multitude of complex methods but for each of the studies undertaken in this work a straightforward function of average concentration and population numbers affected has been used. In order to represent the influence of commuting on exposure the function must be applied on a temporal basis throughout the day in order to capture the variations in air quality and population distribution that exist. The essential exposure function can be represented as:

$$\bar{c}^c = \left(\sum_{i=1}^{n} t_i \frac{\sum_{g=1}^{m} c_{g,i} \cdot p_{g,i}}{\sum_{g=1}^{m} p_{g,i}}\right) / 24 \qquad (4.1)$$

$\bar{c}^c$ **…** population and time weighted average concentration taking into account commuting of population (for a given territory divided into *m* grid cells/polygons)

$c_{g,i}$ **…** average pollutant concentration for time interval *i* at place/grid cell *g*,

p_{gi} **…** number of population at place/grid cell *g* and at time interval ***i*,**

t_i. **….** length of time interval *i (*hours*)* $(\sum_{i=1}^{n} t_i = 24h)$**.**

The commuting population and time weighted annual average concentration $\bar{c}^c$ may then be compared with the static population weighted annual average concentration $\bar{c}$ calculated from the annual average concentration at each point:

$$\bar{c} = \frac{\sum_{g=1}^{m} c_g . p_g}{\sum_{g=1}^{m} p_g} \qquad (4.2)$$

c_g **…**annual average pollutant concentration in grid cell *g*

p_g **…**number of population at place/grid cell *g*

There is discussion as to the chosen metric representing exposure. The standard method of calculating exposure is to represent it as a population-weighted mean concentration as described in (Equation 4.2) which can then be directly applied to concentration response functions. The London study has also presented exposure in units of μg m^{-3}.people. The reason is that it was felt the population-weighted concentration may not capture sufficiently the variations in population.

Time-activity patterns: the time periods used in assessment

The relevance of human activity patterns to determining exposure was discussed previously and some examples of differing patterns in Europe were given. Appropriate time-activity patterns, therefore, will be geographically specific, and in this report the time periods, t_i, used by the three studies varies. For London and for Athens each hour is determined separately giving 24 equal time periods during the day. For Moravia-Silesia each day has been divided into four distinct time periods, being night: 1800-0500 (46%); morning travel: 0500-0700 (8%); daytime: 0700-1600 (37%); and evening travel: 1600-1800 (8%). One further distinction between the studies is that for London and Athens 24 hourly average concentration fields have been estimated representing the annual average concentrations at each hour, whilst for Moravia-Silesia the average concentrations during the four daily time periods have been estimated separately for each quarter during the years, giving a total of 4 time periods x 4 quarters = 16 average concentration maps total.

Estimating health impact – deaths brought forward/premature deaths

One further step is to translate the estimated exposure into estimated health impact. In the case studies undertaken for this study the measure of health impact has been restricted to mortality, interpreted as premature death, or deaths brought forward.

The London study has utilised the definition of 'deaths brought forward', as described by the Committee on the Medical Effects of Air Pollutants (COMEAP) in order to maintain consistency with similar assessments already performed for the Department for Environment Food, Rural Affairs (DEFRA). The notion is the standard that an air pollutant (in this case PM_{10}) is not necessarily the sole cause of a death but can bring forward the death of someone who is already seriously ill (Department of Health UK, 1998). It has been calculated as:

Deaths brought forward = $E * MR * CR$ *(4.3)*

Where:

E …. exposure (µg m^{-3}.people)

MR …. baseline mortality rate (taken as 989.7 per 100,000 people)

CR …. concentration response coefficient.

The value for CR expressed as a percentage change in the baseline mortality rate for a specified change in pollutant concentration, has been taken as 0.75% per 10 µg m^{-3} for PM_{10} as recommended by the UK's Committee on the Medical Effects of Air Pollution (Department of Health UK, 1998).

It should be recognised that applying the COMEAP recommended concentration response coefficients in this way is perhaps not ideal. These coefficients were derived for longer-term exposure on a daily basis and applying them on an hourly basis to derive hourly exposure is a technical inconsistency. However, in the absence of any recommended short-term concentration response coefficients, this was taken as the best available way to derive an hourly health impact result.

Comparison with Silesia study (2008): Included in this report for ease of reference are the estimates for premature deaths arising from air pollution in Silesia, calculated during the 2008 study. The concept is the same as above, but with slight modification and the relative risk in both studies differs strongly (i.e. London: 0.75% per 10 µg m^{-3}; Silesia: 4.3% per 10 µg m^{-3}).

Premature deaths (mortality), *mi*, in a grid cell *i* attributable to PM_{10} concentration, *ci*, over 'non-anthropogenic' background concentration of PM_{10}, *cb,* was calculated according to Equation *4.4* (Ostro, 2006):

$$mi = (ci-cb)*0.1*RR*MR*0.001*popi \qquad (4.4)$$

MR mortality rate per 1 000 inhabitants (10.1 for the Moravian-Silesian region);
RR relative risk 4.31(2.6–6. 1) % to the overall mortality (Künzli *et al.*, 2000) per 10 $\mu g.m^{-3}$
c_b 'non-anthropogenic' background concentration of PM_{10} (5 or 10 $\mu g.m^{-3}$);
pop_i population in a grid cell *i.*

The number of premature deaths (NPD) per million attributable to PM_{10}, was then calculated according to Equation *4.5*.

$$NPD = \frac{1.10^6 \cdot \sum_{i=1}^{N} m_i}{\sum_{i=1}^{N} pop_i} \qquad (4.5)$$

Correction for daytime/night time population differences

Some of the exposure arising when commuting is accounted for will be the result of migration into the city, rather than simply intra-city movement. This can be an issue when comparing assessments accounting for commuting with assessments which do not. The increased exposure in the commuting scenario may reflect an inflated population, such that assessments which dealing in absolute measures of impact (e.g. $\mu g\ m^{-3}$.people, deaths brought forward, etc) will naturally appear greater. A crude adjustment can be made to take account of this by scaling results by a factor that describes the difference between the two residential and daytime commuting population grids:

$$F = R_{pop} / (D_{hours} * D_{pop} + R_{hours} * R_{pop}) / 24 \qquad (4.6)$$

where:
R_{pop} is the total population of the residential population grid
R_{hours} is the number of hours in which the residential population is applied
D_{pop} is the total population for the day time population grid
D_{hours} is the number of hours in which the daytime population is applied

Exposure during commuting

The influence on the total health outcome of incorporating exposure during the period of commute itself into a health impact assessment is relatively unknown. Depending on the journey duration and the mode of travel, individuals are exposed to potentially much higher concentrations than they would be for the majority of the day. However the duration of this exposure will typically be no more than a few hours and often less than one.
In practical terms there are difficulties in determining the length of the commute (duration rather than distance). The time of the commute will affect the hours selected for assessing the exposure at home or at work.

There are complexities in determining the mode of transport used. For people operating their own vehicles this will typically be a single form of transport (most commonly a car) but those using public transport often have several modes of transport in their commute such as walking and a bus ride. It is known that a great number of people travelling into London by train also use the London underground service or a local bus service to get closer to their final destination. In order to undertake a manageable assessment there needs to be a representative, primary mode of transport allocated.

It may be argued that an appropriate concentration response function would need to be allocated to each form of commuting transport assessed. It may not be possible to determine meaningful concentration response functions to use across such a short temporal period as necessary to capture a commuting journey. There are also very specific exposure factors for each mode of transport. A car or bus is largely enclosed, although ventilated, during the journey, affecting the concentrations inside that the passengers are exposed to whereas this is not the case for cyclists, motorcyclists, and walkers. A cyclist is undertaking physical activity during the commute which increases their lung activity and therefore the influence that air pollution will have on them. Someone waiting for the London Underground platform may be exposed to poorly disperse higher concentrations. This would suggest a cohort approach to exposure assessment, on the lines of the 1998 Oslo study.

However, moving from a relatively small city size such as Oslo to a major conurbation such as London or Athens poses issues of data availability and pragmatism which remain to be properly addressed. Thus, the approach adopted for the case studies in this report is from the basis that in general the added impact of the commute itself will be on the low side. For the London, Athens and Moravia-Silesia studies the personal exposure of small cohorts of individuals has not be done; rather city-wide distributions for resident and daytime populations are used to represent defined time periods. This study is limited to the influence of commuting based on home and work locations rather than addressing the influence of the commute itself on exposure.

Nevertheless, it is evident from the discussion above that informed generalisations need to be made to allow such a complex but potentially significant contribution to exposure to be accounted for in future

3.1.4 Availability of commuting data for Europe

There are many sources of commuting data in the European countries which collect the data at country level. However, only few statistical actions on the field of commuting 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.

The main international survey offering the commuting data is the Labour Force Study (LFS) maintained by Eurostat. LFS is 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 at the complete cover of population. The aim of the LFS is to provide time-series with the quarterly results, which application is a good reason to support data for the base data from other sources. The base data should be chosen aiming a good spatial dimension which covers whole population and with very fine spatial resolution. Another disadvantage of the LFS is in the level of spatial resolution of the results. The data can be purchased for most of the countries at NUTS2 level only and for

Germany, Great Britain and Austria even for the NUTS1 only. The prices for the data for the whole set of 26 countries depend whether the whole time series from 1987 (€ 8,000) or a single year (€ 2,000) is requested.

Finding the international data with good spatial cover seems to be a difficult task. Full population cover with good spatial resolution (LAU2) is provided by the population censuses. Most censuses in the European countries include the questions about place of work and school as a standard. We have not been able to study all the results from all the European countries population censuses. But for example, on Czech Republic is quite difficult to find results suitable for describing the commuting process on the very fine spatial level (e.g. LAU2), even for the single country census. However, if we can get the results from each latest census for the European countries, we could get the best cover of the commuting population on the European level with rather comparable data.

The main disadvantage of this census-based approach for commuting on the European level is the date of the last census; the cycle of the census is usually ten years. Another disadvantage would be to handle with data sources at country level. The results should be compatible for all countries if we assume that most censuses use similar questioning about the daily commuting to work or school.

The latest Czech population census (held in 2001) gives commuting data sorted to LAU2 level. The number of people commuting daily to work or school sorted by the municipality of the working place or school could be found in the public database of the Czech Statistical Office. The number of people commuting daily to their working place or school from the municipality of their residence sorted by the LAU2 of their residence was obtained from the Department of regional studies of the Czech Statistical Office afterwards. The daily commuting process in spatial dimension could be constructed from these two pieces of information and the population data. The population for each municipality could be adjusted with the number of people leaving and entering. The next Czech population census should come in 2011, but it is not clear if the census will be fully compatible at the European level. In 2001, people were questioned on the place of their permanent (legal) residence. Commuting data may be affected by the difference between their permanent and usual place of residence. In 2011, according to the experts from Czech Statistical Office, people will be asked on their daily commuting from the place of their usual residence instead of the permanent residence. This change in the questionnaire will provide more realistic commuting data.

3.2 Case Study Findings

This section summarises findings from each case study. Full descriptions are in Appendices.

3.2.1 London: commuters in a very large NW European conurbation

Population: A brief overview of the CommuterView data used to compile residential and daytime populations serves well to describe London from a population perspective (Subregions shown in Figure 3.1):

- In 2001 over 3.7 million people travelled to work in London. Around 2.8 million of these travelled within London, and over 2 million travelled out of their borough of residence.

- Around 722,000 people commuted into London from outside. Over 350,000 of these travelled into Central London, so that including London residents, over 1.5 million commuted to Central London. Thus, around 23% of Central London workers were resident outside London.

- Over half of London's population aged 16-74 lived and worked in Greater London. Almost 9% worked mainly from home (286,000), and only 4% worked outside Greater London, though this represented 236,000 people.

- Most commuters into London came from the South East (374,829), and East (283,750) regions. The South East was the most usual external work destination for London residents (131,812). Commuting into London for work exceeded commuting out by a factor of over three.

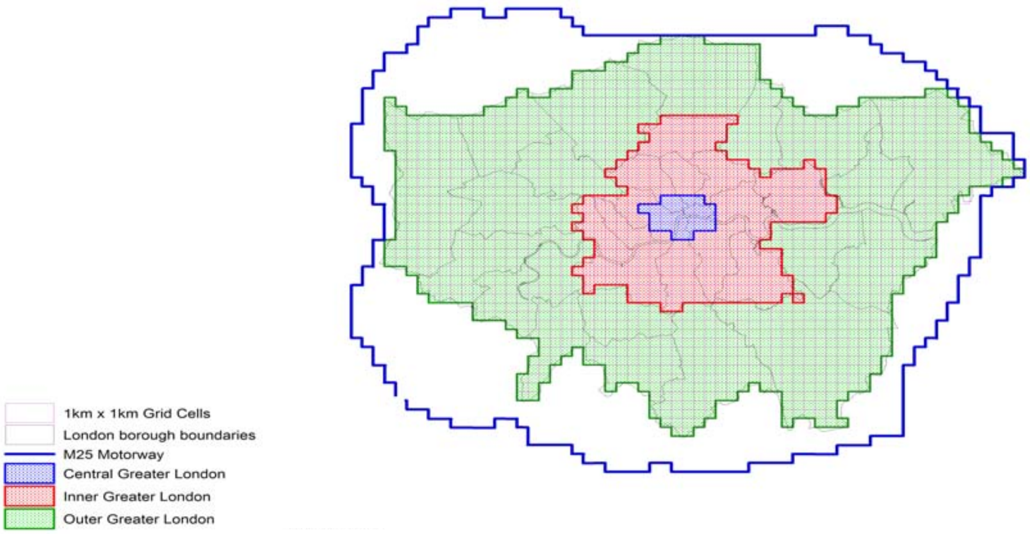


Figure 3.1: Greater London, and division into Central, Inner and Outer regions. M25 Ring road shown in blue for ease of geographical reference.

<u>*Air quality:*</u> As a baseline, the map below from the AEAT PCM model shows estimated average annual concentrations of PM10 (µg m^{-3}) for Greater London and beyond (Figure 3.2), covering all SE England. The M25 ring road is shown in magenta for ease of geographical reference.

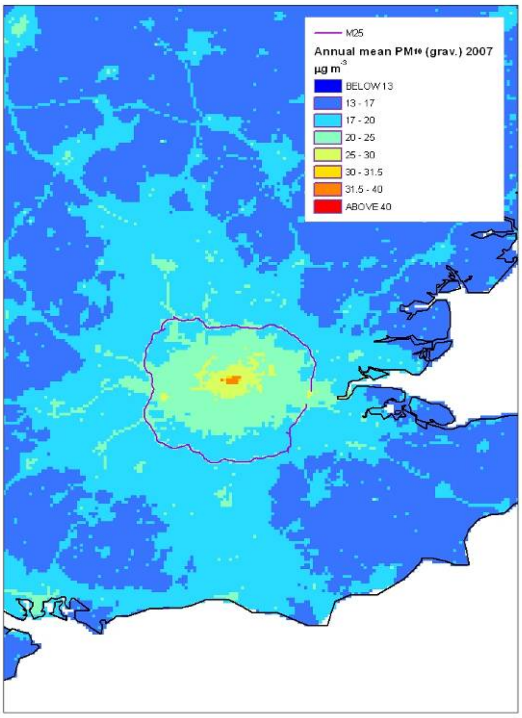


Figure 3.2: Annual average PM10 concentrations (µg m-3) in Greater London and SE England in 2007.

Running this model to derive annual averages for each hour separately during the day reveals the first result of this work, which is the dual peak in PM_{10} concentrations, coinciding with the morning and afternoon commutes. Figures 3.3 (a-e) show particularly low concentrations in the city at midnight (a), which rise until 08.00 (b) as a result of commuting road traffic emissions movements to work. Concentrations then dip during the daytime as shown for 13.00 (Figure 3.3c) and then rise to maximum concentrations for the day around 17.00 (Figure 3.3d), again due to the road traffic emissions from the journey home. The concentrations decline later in the evening (Figure 3.3e) as traffic subsides.

The pattern, therefore, generally conforms to the expectation that commuting populations travel from lower concentration regions to high, thus that daytime concentrations are elevated during the period when population will be highest. However, the daytime dip in concentrations suggests that daytime exposure may be less elevated than expected.

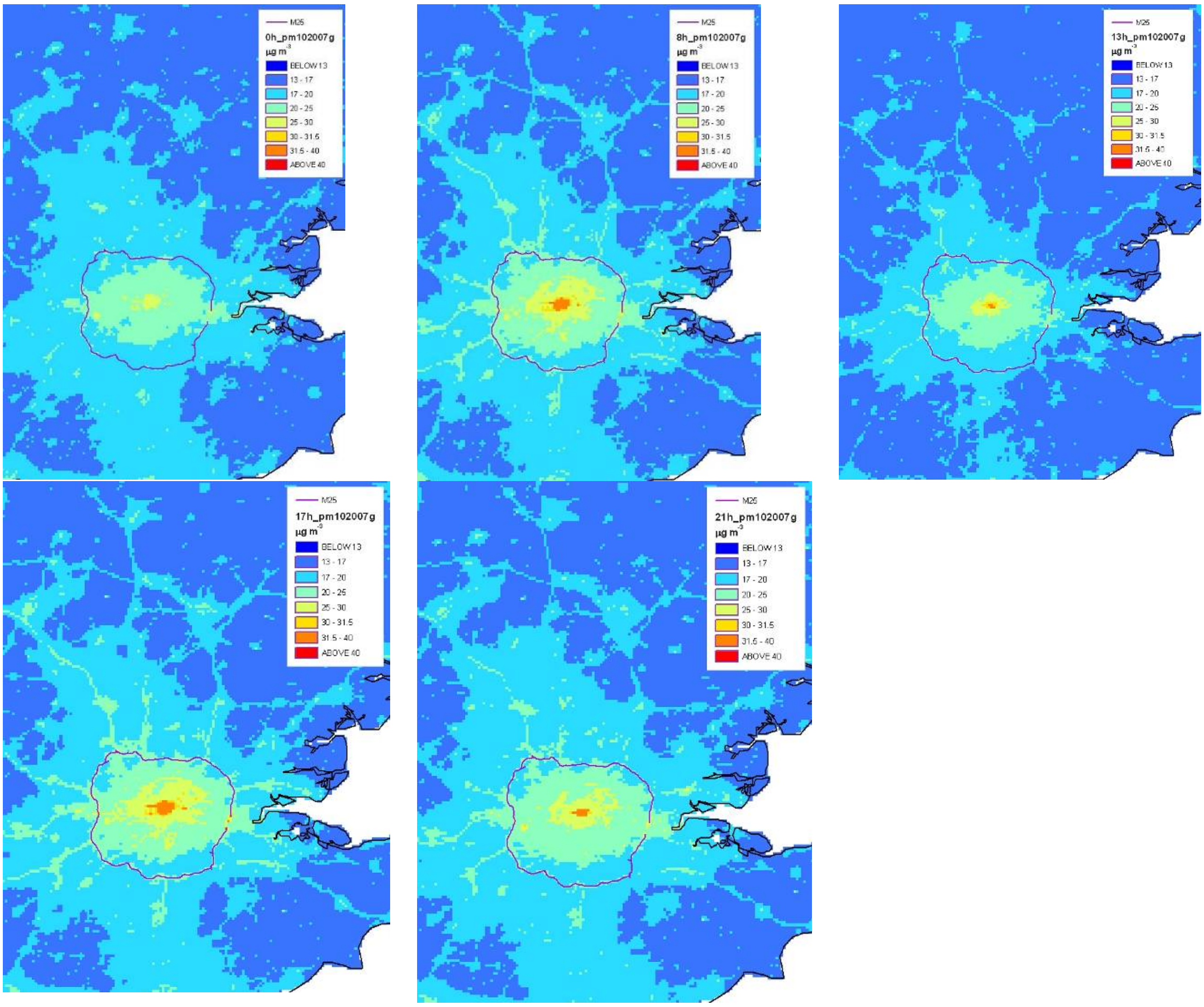


Figure 3.3: Hourly air quality maps for London – PM10 (µg m^{-3}) (2007).

Exposure: Exposure has been estimated and aggregated for three subregions of the Greater London area, these being Central, Inner and Outer London. These are shown in Figure 3.4, with population and exposure data in Table 3.2 and 3.3. The total Greater London population in the daytime is larger than the residential population by almost half a million people (487,465). Thus, 6.4% of the daytime population are not accounted for in the residential population grid. Careful interpretation of exposure changes is needed, as assessments accounting for commuting may increase apparent exposure more than would be the case if these extra people were captured in a residential assessment. Expanding the domain to a very wide area to the west, south and north of London may be an appropriate future measure. On the converse side, a much larger domain divides the whole UK into a small number of large regions which may not aid detailed work, and a large influx to high concentration regions will be from low concentration districts beyond Greater London. Thus, the impact of the commute may be greatest upon a cohort missed by residential assessments.

Increased exposure results from daytime concentrations being elevated during the period when the population is highest, but with the commute period being the period of maximum impact. The London study did not intend to examine commuting exposure per se, for example in the way the 2008 Oslo study did. However, in support of a robust baseline analysis the subject of exposure during the period of commute was given previously.

Table 3.2: Summary of population grids used

Zone	Area (km^2)	*3.2.1.1.1.1 Residential population*	Daytime population
Central London	25	181,805	1,159,870
Inner London	307	2,601,100	2,541,240
Outer London	1,271	4,352,640	3,921,980
Total	1,603	7,135,801	7,623,266

Table 3.3: Exposure summary (µg m-3.people)

Zone	Exposure (µg m^{-3}.people)	
	Non-commuting	Commuting
Central London	5,575,670	18,995,000
Inner London	65,489,900	65,259,900
Outer London	98,652,900	94,693,600
Total	169,718,470	178,948,500

Redistribution in exposure from outer areas to the centre is observable. The large numbers in Table 3.3 can be hard to compare, and so the same information s given graphically in Figure 3.4. After weighting to account fr daytime-night time population differences, these estimates have then been converted to population-weighted annual mean concentrations for Greater London (Table 3.5), and have also been used to estimate impact upon mortality, described as numbers of deaths brought forward.

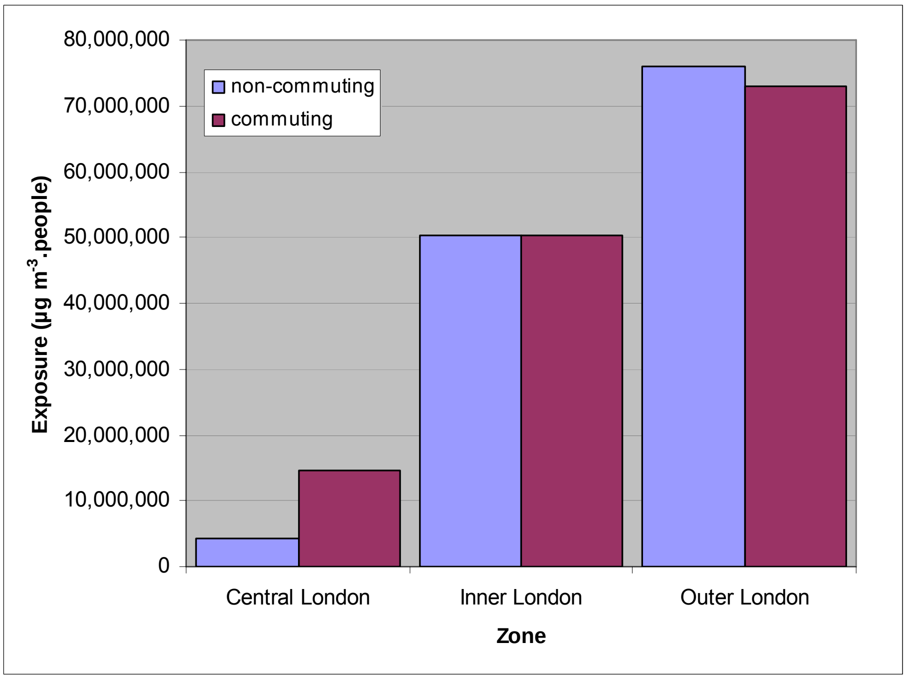


Figure 3.4: Influence of commuting on geographical distribution of exposure

Table 3.4 Population-weighted mean representations of exposure by scenario

Population-weighted annual mean (µg m^{-3})	
Non-commuting	**Commuting**
23.8	24.4

The exposure estimates discussed above have been used to take the assessment through another stage in order to calculate the number of deaths brought forward (Table 3.5). As explained above, on account of the increased daytime population as compared to the residential population, there is an automatic bias towards greater apparent impact. Adjusting for this as described reduces the anticipated numbers of deaths brought forward to below the values in Table 3.5. The adjusted total for the commuting scenario of 969, an increase in deaths brought forward as a result of taking commuting into account of 2.5% compared with the 5.4% calculated without adjusting for this population effect. It should be noted that this is in line with the estimated increase in population-weighted air concentration of 2.5% after accounting for daytime/night time population balance.

The exposure results illustrate that:

- Including commuting in exposure assessment increases net city-wide exposure to PM10.
- The influence is not uniform, but has distinct geographical expression across the city.
- Central London displayed the most marked increase in exposure due to the inflow of commuters.
- Inner London displays no significant change in exposure with inclusion of commuting: the inflow and outflow of commuters are cancelling out.
- Outer London displays a net decrease in exposure due to the outflow of commuters.

- Traditional assessments for residential populations understate exposure in the centre because the central domiciled population is low, yet working populations are high.
- Similarly, such assessments overstate exposure in outer areas as the domiciled population is higher than the working population.
- Concentrations of PM_{10} are lower nighttimes than daytime, thus increasing the impact.

Table 3.5: Number of deaths brought forward

Zone	Deaths brought forward	
	Non-commuting	Commuting
Central London	32	108
Inner London	374	373
Outer London	563	541
Total	969	1022

The London study demonstrates that taking commuting into account increases the estimated exposure to air pollution. Proportionally, this increase was relatively small (2.5% for adjusted estimates of increase in deaths brought forward) but in Central London and Outer London this influence of commuting is notable, reflecting the population shifts from the outer city to the centre during the day. Note again that the London study did not include the exposure during commuting. It is anticipated that this would increase the exposure and the health effects.

Accurately capturing the influence of commuting across city boundaries needs the domain of assessment needs to extend an equal distance, so that the people represented in the city during the working day are also represented outside of working day hours (i.e. they are included in the residential population grid). The exact domain needs to be set to a reasonable and manageable area to ensure that the majority of regular commuting is captured. A difficulty is that whilst this approach may best estimate total impact upon a population, more discrete information is needed for targeting of response. The Oslo study showed potentially far greater impact upon smaller sections of the community, which overall geographical Figures of residence or employment may fail to reveal. In a city the size of London, a small percentage translates into large numbers. If this added impact is focused on a small group, the impact may be entirely misrepresented.

3.2.2 Moravia-Silesia: urban exposure in the industrial heartland

Population: The Moravian-Silesian region is in the north east of the Czech Republic on the borders with Poland and Slovakia located on the historical transport route running north-south from the Baltic to the Mediterranean. The industrial tradition is long, being one of the most important industrial regions in Europe since 19th century. Since 1989 restructuring and revitalization of heavy industry has taken place, with the closure of many coal mines, inflow of domestic and foreign investments, and growth of the automobile industry. The Moravia-Silesian region (MSR) is one of the most polluted regions of both the Czech Republic and Europe. PM10, benzo(a)pyrene (BaP) and benzene ambient concentrations exceed EU limit and target values. Population density is about 230 inhabitants/km^2. The largest city is Ostrava, with 310 741 inhabitants. Almost 62 % of citizens live in towns with a population over 20 000 inhabitants and about 25 % of inhabitants live in settlements with population lower than 5 000.

A baseline population map was constructed for the Moravian-Silesia region at 1km resolution (see Appendix) based on the population registry for January 1st, 2009 collected by the Czech Statistical Office. Population data were converted from the original LAU2 format to a 1x1 km grid on the proportion of built up land in each grid square. Coloured grid-cells represent inhabited areas.
This baseline map should be considered the 'night-time' population map, having no account of daily commuting. Using commuting data a map of commuting balance can be constructed from this. The number of people commuting daily outside from each municipality to their work or school on the LAU2 level is subtracted from the population and the number of people commuting daily inside each municipality is added. The result may be interpreted as a 'daytime' population, reflecting commuting movements. The difference between the 'night-time' population and the 'daytime' population is given by the number of people commuting to a municipality other than that of their residence, minus the number of people commuting out of their residential municipality to work or school. This difference is considered as the commuting balance. Expressed both as absolute numbers and percentages of the residential total, this balance is shown in Figure 3.5.

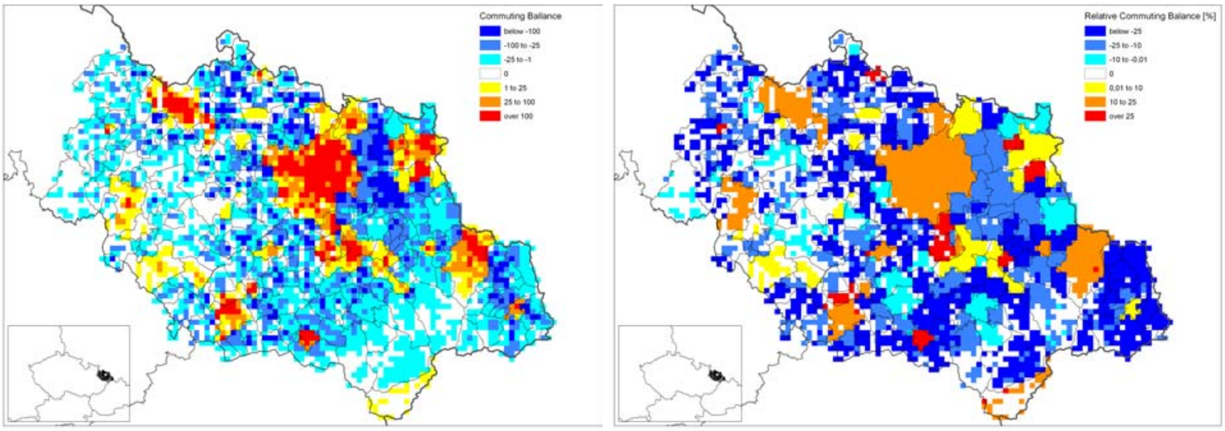


Figure 3.5 Map of commuting balance in Moravian-Silesia region (left: absolute, right: relative)

The result is the population respecting the daily commuting that could be interpreted as 'daytime' population. The difference between the 'night-time' population and the 'daytime' population is given by the number of people commuting inside the municipality different from the municipality of their residence subtracted by the number of people commuting outside of their residence to work or school. This number is the commuting balance.
The geographic distribution of the commuting process could be different from the distribution of built-up areas used for the construction of the population map. This is the significant point of uncertainty in the commuting balance map in the areas inside each municipality. However the sum of each municipality remains correct *according to the statistical data.*

Air quality: Basic input for mapping of PM10 concentrations are monitored data from the national ambient air monitoring network. Observed PM10 concentrations in 2007 and 2008 at all background (rural and urban/suburban) automatic stations in the eastern Czech Republic were used (25 background stations: 19 urban/suburban, 9 rural). Figure 3.6 depicts the

temporal cycle of 3-hour PM10 concentrations averaged across all background stations for winter and summer seasons.
A night time air concentrations maximum is a particular feature of the region. Most marked at background stations, it is also found to a certain degree in industrial and traffic stations (Figure 3.7). Thus, across Moravia-Silesia we see a daily cycle with night time maximum superimposed upon a seasonal pattern with a winter maximum. Elevated night time concentrations in rural areas at levels approaching those of daytime trafficked sites are a particular feature of this region of central Europe.

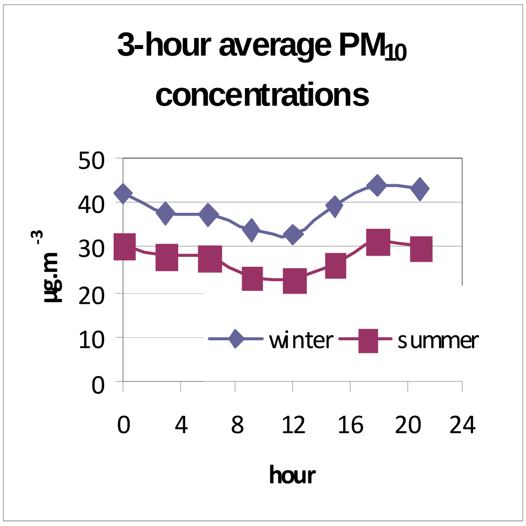


Figure 3.6 PM10 concentrations during the day, for summertime and for wintertime, as observed across all background stations in Moravia-Silesia.

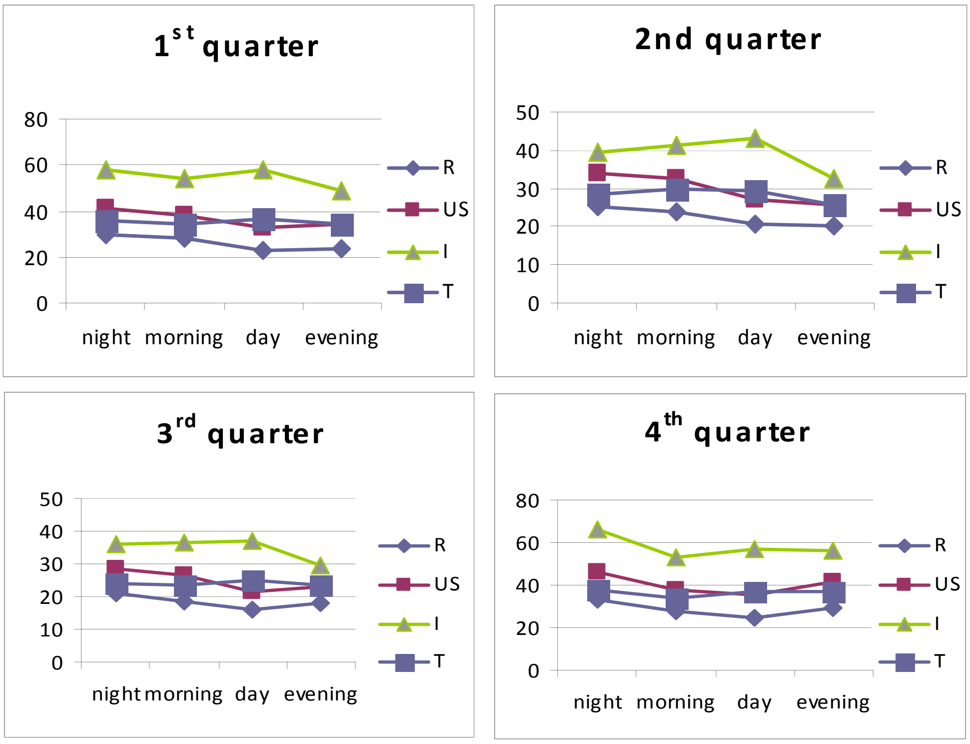


Figure 3.7: Daily PM10 concentrations averaged over indicated intervals (night, morning, day and evening) for each quarter and station type (R: rural, US: urban and suburban, I: industrial, T: traffic)

<u>*Exposure:*</u> Air quality maps were produced for each quarter of the year, and for four periods of the day, namely:

- night, i.e. time when the residents' majority is at home (from 18:00 to 05:00 local time)
- morning, i.e. morning travel time to work (from 05:00 to 07:00 local time)
- day, i.e. time when the majority of the residents are at work (from 07:00 to 16:00 local time)
- evening, i.e. the evening travel time from work (from 16:00 to 18:00 local time).

These four periods were compared with residential and daytime population balance maps to allow estimation of total population exposure, expressed as population weighted PM10 concentrations.

Table 3.6: Overview of population weighted PM10 concentrations reflecting commuting, reflecting effect of commuting and weekends and population weighted PM10 concentrations if commuting is not taken into account for particular quarters and whole year (averages of 2007 and 2008).

	1st quarter	2nd quarter	3rd quarter	4th quarter	year
Population weighted PM10 concentrations reflecting commuting	**40,06**	**33,13**	**27,16**	**44,81**	**36,29**
Population weighted PM10 concentrations reflecting commuting and weekends	**40,03**	**33,10**	**27,14**	**44,77**	**36,26**
Population weighted PM10 concentrations, commuting not taken into account	**39,95**	**33,04**	**27,09**	**44,67**	**36,19**

The results show only a small effect on assessed population weighted concentration (Table 3.6) and premature deaths (Table 3.7) when commuting is taken into account. The obvious reason lies in the daily pattern to PM10 concentrations when highest ambient air concentrations are during the night time, whereas during travelling and working hours concentrations are lower both in residential and urban/industrial areas. The difference in PM10 concentrations between residential and urban/industrial areas is apparently least during the night hours, thus offering limited respite to the residential population.

Table 3.7: Number of premature deaths per million in Moravia-Silesia attributable to PM10 reflecting commuting, reflecting effect of commuting and weekends and number of premature death per million concentrations if commuting is not taken into account for particular quarters and whole year (averages of 2007 and 2008). PM10 'non-anthropogenic' background set on 5 μg.m^{-3}.

	1st quarter	2nd quarter	3rd quarter	4th quarter	year
Number of premature deaths per milion attributal per PM10 reflecting commuting	1526	1225	965	1733	1362
Number of premature deaths per milion attributal per PM10 reflecting commuting and weekends	1525	1223	964	1731	1361
Number of premature deaths per milion attributal per PM10, commuting not taken into account	1522	1221	962	1727	1358

3.2.3 Athens: a high density Mediterranean urban environment

Population: The Greater Athens Area (GAA) lies in a basin adjacent to the sea and surrounded by mountains on three sides. Most industries lie close to the sea and the harbour of Piraeus in the south-west.

National Census data for 2001 from the National Statistical Service was used, revealing a monocentric urban structure with a residential population of around 4.3 million living in 82 municipalities. This area has the largest population, the highest density and the highest economic activity in the country. Around 25 and 30% of the population lives and works in the centre, respectively. Applying population flow data to this gives the difference between the residential population, the number of people entering a municipality, and the number of people leaving it, in effect giving a population balance.

The results for the population distribution within the Attica region and the daily population variation due to commuting to work, show that most of the commuting in GAA takes places place within the main Athens – Piraeus area and is mostly directed towards the Athens city centre, with a relatively small influx of people from outside the GAA. More specifically, of the GAA total population of around 4.3 million, around 3.4 million reside in Athens and Piraeus. During the day around 240 000 people enter the GAA, which corresponds to an increase of the daytime population of GAA of approximately 5%. Figure 3.8 shows the daily variation of the population distribution in the GAA due to commuting for work.

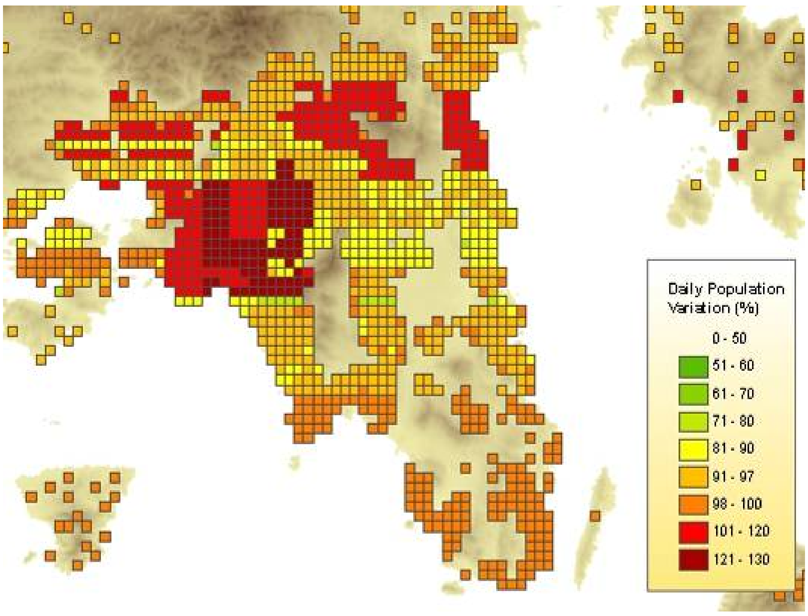


Figure 3.8: Daily percentage variations in the population distribution in the Greater Athens Area (GAA) due to commuting, at 1km resolution.

Air quality: Air quality across the model domain was extracted at 1km resolution and a total of 24 annual average concentration maps produced for each hour of the day. These reveal a nightime minimum to air concentrations of PM10, and a peak during the morning beginning coincident with the morning commute (Figure 3.9).

The greatest daily variation is focused on the Athens-Piraeus area, a fact which is better revealed by looking at the hourly change in average concentrations across the central Athens-Piraeus area (Figure 3.10 - 50km resolution grid) and across the remaining region (Figure 3.10 - 150km resolution grid). The central region experiences higher concentrations, greater variation and a peak in the early morning commute period.

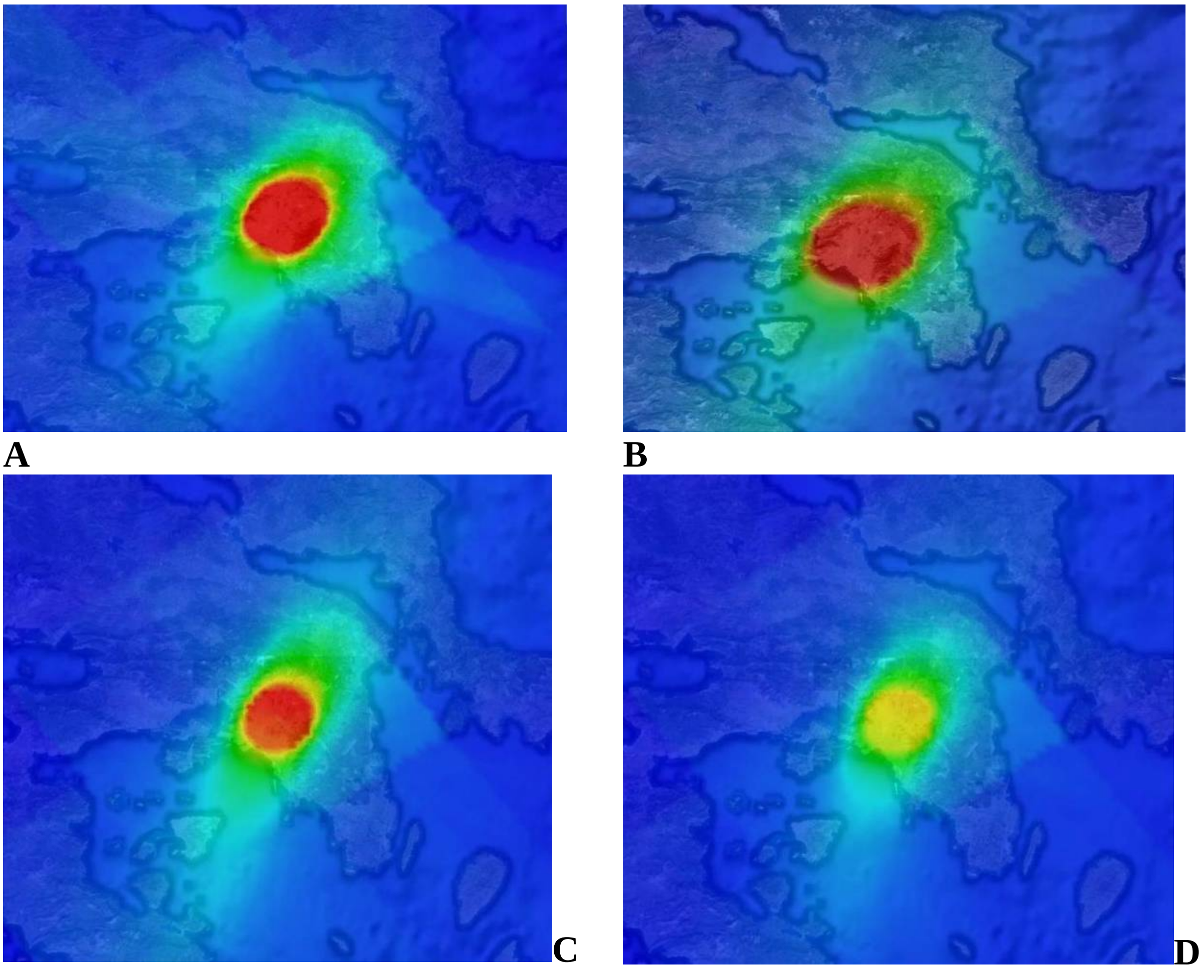


Figure 3.9: Annual average PM10 selected hours; A: 08:00; B: 11:00; C: 15:00; D: 19:00

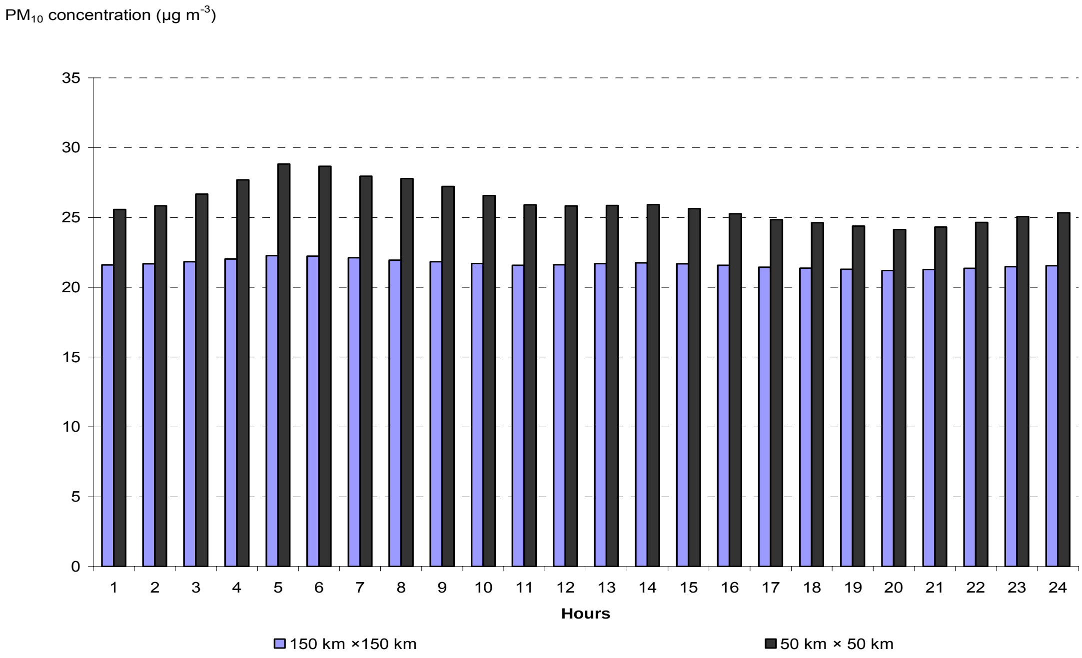


Figure 3.10: Daily variation of the average PM10 concentration level, comparing the whole GAA (150km x 150km) with the Athens-Piraeus area (50km x 50km)

<u>*Exposure:*</u> The Athens – Piraeus area is responsible for the greatest part of the population exposure in the GAA both for the commuting and the residential scenarios. The reason is that both the PM_{10} concentration levels close to the city centres of Piraeus and Athens the population density are considerably higher compared to the surrounding areas within the GAA. Moreover, a comparison of the population exposure results between the commuting and the residential scenarios reveals that commuting towards the working place has a relatively small contribution to the overall estimated exposure across the GAA (≈3%). Table 3.8 gives the comparative exposure under residential (non-commuting) and commuting scenarios. As the large numbers can be hard to compare, the same information is given graphically in Figure 3.11.

Table 3.8: Exposure summary (μg m^{-3}.people) for the Greater Athens Area

Area	**Exposure (μg m^{-3}.people)**	
	Non-commuting	**Commuting**
Athens - Piraeus	104,923,567	105,064,476.7
Surroundings	18,811,398	22,595,411.3
Total GAA	123,734,965	127,659,888

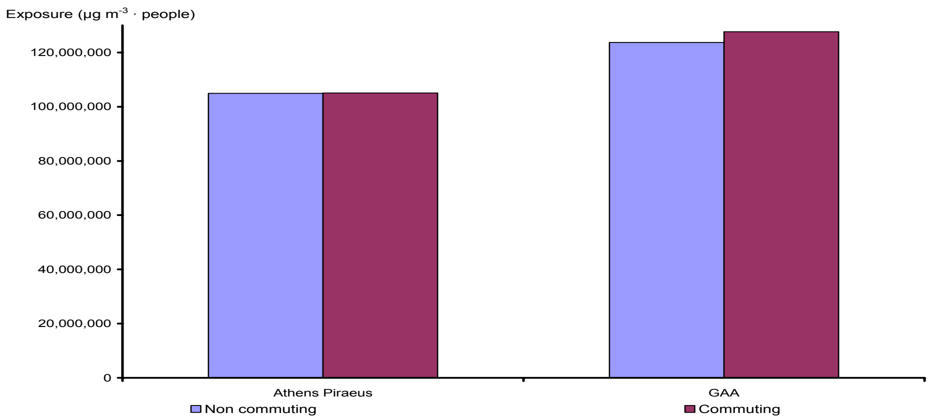


Figure3.11: Total exposure for the commuting and static population scenarios in the GAA

On the basis of the population-weighted annual mean concentrations exposure was converted to excess death from chronic PM_{10}. The increase in excess deaths was found to be of the order of 4%, with the largest part occurring within the Athens-Piraeus area (Table 3.10 and Table 3.11).

Table 3.10: Population-weighted mean annual concentrations

Population-weighted annual mean (µg m^{-3})		
Area	**Non-commuting**	**Commuting**
Athens - Piraeus	29.6	30.9
GAA	28.8	29.0

Table 3.11: Estimated numbers of deaths brought forward for the GAA

Deaths brought forward		
Area	**Non-commuting**	**Commuting**
Athens - Piraeus	777	809
GAA	916	954

4 Influence of spatial scale

4.1 Spatial Scale Resolution in Exposure Modelling

The importance of spatial scale resolution in exposure modelling was presented In Chapter 2; “Review of Exposure Assessment Method”. Spatial scale resolution may affect the outcome of exposure assessment and in particular, the use of coarse grid may involve exposure underestimations. Barrett *et al.* (2008) reported the results obtained in case studies performed in Athens, central London, and Silesia aiming to evaluate the comparison between assessing the health impact of air pollution at city scale and at neighbourhood scale. The example of Silesia (presented in Chapter 2) is summarized below.

4.2 Influence of spatial scale in the case studies

Summary of the 2008 estimates for Silesia.
The first estimates of the effect of the spatial scale of assessment upon estimated health impacts of air pollution were presented in the 2008 report, concerning Silesia (MSR). For convenience and as a suitable starting point, the conclusions are presented here as Table 4.1.

Table 4.1: Population exposure to PM10 annual average concentration in 2006 in MSR

Grid size [km x km]		*Concentration classes [µg.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 µg.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 µg.m^{-3}
	% of pop.:	**0.0%**	**0.6%**	**12.6%**	**43.3%**	**12.3%**	**31.2%**	

Population exposure analysis was done at a 10x10 and at 1x1 km grid size to estimate influence of mapping grid size on estimated population exposure. The estimated population exposure is higher for the finer grid. With increasing grid size the estimates of population exposure are evidently more biased towards higher exposures. Within a grid cell of the larger size the areas with high concentrations gradients are ‘smoothed’ by cleaner areas which have lower population density, thus reducing the spatial peaks. Low concentration / low population areas (typically less built up, more rural) effectively ‘dilutes’ the influence of smaller higher population/concentration areas, resulting in lower exposure estimates.
It was concluded that accounting for more detailed geographical scale (1x1 km grid) will increase the estimated exposure of the total urban population on a 3.26% as compared with estimates based on a total urban population exposed to a single averaged air pollutant concentration (10x10 km grid).

The impact of this exposure upon health was estimated in terms of numbers of premature deaths, and the consequences of spatial scale for numbers of deaths were estimated as given in Table 4.2.

Table 4.2: Number of premature death per million in Moravia-Silesia attributable to PM_{10}

PM_{10}	Grid size *[km]*

'nonanthropogenic' background	1x1	10x10
5 µg.m^{-3}	**1573.3**	**1514.8**
CI	951.3-2231.9	915.9 – 2148.9
10 µg.m^{-3}	**1356.1**	**1297.72**
CI	820.01-1923.9	784.6 – 1840.9

The assessment of air pollution impact on health of the Moravia-Silesian population, and especially for the Ostrava city, carried out by the Institute of public health estimated a shortening of average live expectancy of about 2 years, whereas for the Czech Republic as a whole the shortening is about 1 year (Šebáková *et al.*, 2008). Life expectancy in this region: men 71.86 (73.45 in the Czech Republic) years (a difference 1.6 years) ; women 78.84 (79.67 in Czech Republic).

Spatial scale influence on exposure in London

A similar comparative assessment has now been conducted for London. The assessment results have a spatial resolution of 1x1km - the highest resolution at which the air quality model is capable of providing PM_{10} concentrations.

To determine the consequence of moving to a coarser resolution of 10km (equivalent to current Topic Centre evaluations, and consistent with the Silesia evaluation), the 1x1km data for London was aggregated to a 10 x 10km resolution and the exposure recalculated. At this resolution comparison of the separate subdistricts of Inner London, Outer London and Central London becomes problematic; therefore results are presented for Greater London in total.

Table 4.3 summarises results of the exposure assessment at the coarser resolution for each assessment scenario (commuting .v. static populations) and the difference in result compared with the higher resolution assessment. The percentage decline from the 1x1km results is also given. In both scenarios the exposure was lower for the coarser resolution, reinforcing the conclusion from the Moravia-Silesia study that lower resolution assessments result in an under-estimation of the public health impact.

It is striking that the estimated percentage decrease in the estimated numbers of deaths brought forward for London with a decrease in resolution from 1km to 10km is very close indeed to the estimated decline in numbers of premature deaths in Silesia with the same change in spatial resolution.

Table 4.3 Comparison of estimated exposure at 1x1 km and 10x10km resolution in London

Scenario	**1x1km**		**10x10km**		**% change from 1 to 10km resolution**
	Exposure (µg m^{-3}.people)	**Deaths brought forward**	**Exposure (µg m^{-3}.people)**	**Deaths brought forward**	
Non-commuting	169,718,470	969	164,949,552	942	- 2.81
Commuting	178,948,500	1022	171,242,944	978	- 4.31

*** Exposure in this Table presented only using gravimetric PM_{10} concentration, not TEOM equivalent used to calculate the deaths brought forward**

5 Discussion and Conclusions

5.1 Discussion

This report has presented a short overview of the state of the art on exposure science and recent results on improved exposure assessments at the urban scale, including the case studies performed for London, Silesia and Athens. 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.
Different methods are used in exposure assessment but not all of them are equally suitable for urban exposure estimates. Table 5.1 summarizes the exposure assessment methods commonly used and their main advantages and disadvantages for health studies. Given the current state of the knowledge, hybrid models in where the strengths of different methods are combined constitute an important tool for reducing exposure estimates uncertainties. The hybrid models as indicated in Figure 5.1 may result from the combination of models based on principles of physics and chemistry (Mechanistic models) and models based on statistical relationships (Empirical models). However, the uncertainties associated with the modelling will be transferred to the concentration field outcome. In addition variability of the concentration field at urban scale constitutes a critical factor which has been intensively studied. The uncertainties associated with the methods and the variability observed at urban scale represents important challenges for performing accurate exposure assessment.

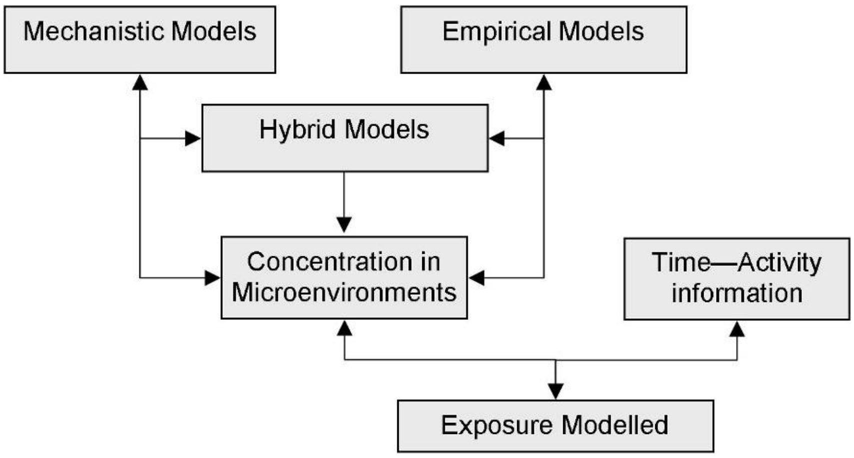


Figure 5.1: Schematic diagram of models used in exposure assessment (After NCR, 1991; WHO/IPCS, 2005).

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 and briefly summarized in this report. Current exposure assessment estimates are commonly based on static residential address which may involve high uncertainties. An increase of 50 and 100% 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 exposure estimates obtained in the case study in London are consistent with results from previous studies and demonstrate that taking commuting into account increases the estimated exposure to air pollution. On net, this increase was relatively small at 2.5%, but in Central London and Outer London this influence is notable and reflects the significance to exposure of population shifts from the outer city to the centre during the day. Low increases have also been reported in the case study in Athens (≈4%) and in other published studies (8% increase;

Gulliver and Briggs, 2005). In the case study performed in Moravia-Silesia, only a small effect was observed on assessed population weighted concentration when commuting is taken into account. The reason lays in the PM10 daily pattern concentrations; highest ambient concentrations are observed during the night, whereas low concentrations are observed during travelling and working hours.
The case studies and the literature review points to the need for additional research on exposure assessment methodologies, and comparison of different exposure estimates may contribute to the better understanding of exposure estimates errors.
Due to commuting between home and work pace, the population number within cities undergoes large variations. Double and triple population during working hours have been suggested for city centres 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 2.4). These populations' shifts may involve important variations in exposure estimates compared to those static-based estimates. Thus the 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. However, how to deal with time activity information varies between studies, some of them consider data from time-microenvironment-activity diaries (TMA diaries) or dynamic transport models, others follow example individuals, compare static home population and static work population, whereas others consider static home population with statistical description of the variations. Consequently more research is needed on the analysis of mobility or time-activity-space patterns. As it was suggested by Jerrett *et al.* (2005), collaboration with transportation experts may contribute to develop more accurate exposure estimates. An example of this collaboration is the study performed by Beckx *et al.* (2009) which combine an air dispersion model with "A Learning-Based Transportation Oriented Simulation System (ALBATROS)" to predict activity-travel patterns and obtain a dynamic exposure assessment.
Pollutant concentration field experiments, such as carried out in the case studies, moreover show significant concentration variations within cities. These variations are associated with factors such as wind directions, morphometry of the city / neighbourhood and pollutant dispersion. Spatial resolution and detailed characterization of the pollutant concentration field may reduce uncertainties and improve the outcomes from exposure estimates. The work performed in Moravian-Silesian region (Barrett *et al.*, 2008) constitutes an excellent example. Modelling the population exposure 1x1 km grid size increases on a 3.26% with regards to the exposure modelled based on 10x10 km grid size. These results are reinforced by the study performed in London, which provide more evidence that higher resolution exposure studies increase the exposure estimate.

5.2 Concluding remarks

- Different methods are used in exposure science to estimate exposure of intraurban populations (Table 6.1). Some of the methods involve high uncertainties whereas others such as hybrid models combine the strengths of different methods.

- Exposure estimates depend on the concentration of pollutants of concern and on the population. Therefore significant variations in one or both factors will influence the estimates.

- Urban population distribution undergoes important daily variations during day due to commuting to and from work and other commuting. Noteworthy are the increase reported for cities as Oslo (Norway; *100%* increase; Municipality of Oslo, 2007), Lisbon (Portugal; *100-200%* increase; Borrego *et al.*, 2006), Haarlemmermeer (The Netherlands; 44% increase; Vos and Trijssenaar, 2000), Utrecht (The Netherlands; 31% increase; Vos and Trijssenaar, 2000), Washington D.C. (USA; *71.8%* increase; U.S. Census, 2000), Irvine city (USA; *73.83%* increase; U.S. Census, 2000) and Salt Lake City (USA; *72.18%* increase; U.S. Census 2000).

- Daily commuting time to and from work is significant, between 0.72 and 1.70 hours day^{-1}.

- People spend approximately between 20 and 30% of their time away from home (e.g. work, school) and 4-5% of their time in transit between different microenvironment, commonly commuting from and to work during peak traffic hours.

- Commuting elevate personal exposure to levels above those modelled from home addresses or census tracts and increases of 50 and 100% have been reported in different studies (Özkaynak *et al.* 2008; Pérez Ballesta *et al.* 2008; Barrett *et al.*, 2008). However, lower increases have been observed, for instance in the case studies in London and Athens. But the influence of commuting is notable in some areas and reflects the significance to exposure of population shifts from the outer city to the centre.

- The absence of human activity information in exposure modelling, and in particular commuting to and from work may generally underestimate exposure to air pollution in urban areas.

- The availability of data on commuting patterns (commuter datasets) is essential for performing exposure estimate analysis.

- Pollutant concentration field experiments show significant variations within cities. Spatial scale and detailed characterization of the pollutant concentration field may reduce uncertainties and improve the outcomes from exposure estimates.

- Detailed geographical resolution may increase the estimated exposure of total urban population. Increase of 3% has been reported from exposure estimated at 10x10 to 1x1 km grid size.

- Additional research is needed to quantify exposure estimates variations due to taking into account factors such as spatial scale and commuting into the exposure assessment method.

5.3 Further Work

- Continued efforts to identify and source new or updated data to improve the currency and quality of the commuter datasets is recommended for further work, though it is recognised that London already has a higher quality and volume of data than many European cities may be able to acquire for similar assessments.

- Further work is recommended to focus on distance travelled to work and method of travel-to-work of commuters, with particular focus on commuting into Central London, in particular the City of London and Westminster (the two boroughs with the largest differences in resident and daytime populations, including highest daytime population densities that might significantly impact on the outcome of exposure studies).

- Efforts to incorporate a representation of exposure during the commuting journey itself would ideally need to address the start and end points of the journey to permit a calculation of the distance involved. If this was unmanageable a potential alternative approach might be to establish a series of cordons (rings) radiating from London city centre and analysing movements between them. The cell by cell matrix approach (though ideal) is not possible with the CommuterView data set however, the more generalised cordon approach is a more appropriate alternative possibility. A third possibility might be to use additional data outside the CommuterView data set to infer information about peoples' journeys and mode of transport.

- Information about the mode of transport involved would need to be coupled with journey duration. It is envisaged that representing complex journeys in an exposure assessment would be unmanageable on a city-population scale and so a commuting journey would be assumed to involve a single, primary mode of transport. Information about the journey distance and mode of transport could then be used to estimate a journey duration time via assumptions about speed based on the primary mode of travel. Mode of transport data exists in the CommuterView dataset but is not directly related to the population movements described by the dataset. However, generalisations about the mode of transport could be made or there may be appropriate alternative data sets in existence that may help with this.

- Some research into typical concentration ranges associated with different modes of transport would be a necessary component for commuting journey exposure – a literature review of relevant studies might be a suitable method to obtain this information.

- Concentration response functions are not generally defined with such short-term exposure in mind or for such specific environments as those associated with commuting journeys. On order to gauge the effect on exposure, these concentrations are not strictly necessary but in order to calculate a health impact result they are. In the absence of specifically developed concentration response relationships it is proposed that existing relationships are used.

- Future developments in this case study area would benefit from additional analysis of commuter flows across three Government Office regions, specifically London, the South East and the East of England (these three regions are the major commuter movements) in order to provide a holistic view of commuters who travelled into and out of London by region and districts. This would prevent an unreasonably high apparent increase in exposure due to commuting from outside London as noted in the analysis.

Table 5.1: Evaluation of air pollution exposure assessment methods (After Jerrett et al., 2005; Zou et al., 2009)

METHOD / DESCRIPTION	ADVANTAGES	DRAWBACKS	EXAMPLES
Direct Exposure Assessment Methods			
Personal Monitoring **employs devices to register pollutant concentration of individuals while they move between different microenvironments.**	• Direct measurement • Accurate estimate of personal exposure	• Labour-intensive, time consuming, costly • Limitations for population exposure assessment.	
Biological Monitoring **(Biomarkers) involves the analysis of body material to determine pollutant concentrations levels.**	• Measure the internal dose of a pollutant in human body • Can improve the accuracy of exposure assessment	• High cost and time consuming • Hard to differentiate between exposure pathways and chemicals	Topinka *et al.*, 2007; Romieu *et al.*, 2008
Indirect Exposure Assessment Methods			
Proximity Models **measure the proximity of a subject to a pollution source**	• Simple • Useful as exploratory analysis	• Exposure misclassification • Controversial conclusions	Hoek *et al.*, 2002; Brender et la., 2008;
Interpolation Models**: estimate pollutant concentration field based on geostatistical techniques and monitoring stations distributed throughout the study area.**	• Use pollution measurement data	• Require a dense network of sampling locations	Pikhart *et al.*, 2001; Ritz *et al.*, 2000;
Air Dispersion Models**: estimate pollutant concentrations over space and time by numerical processing which incorporates emission data, topography and meteorological data.**	• Provide complete pollutant concentration profiles in space and time in areas without sufficient monitoring network • Can provide high-resolution analysis of patterns in health outcomes and environmental	• Costly data input • Need for extensive cross validation with monitoring data.	Clench-Aas *et al.*, 1999; Bartonova *et al.*, 1999; Nyberg *et al.*, 2000.

	factors		
Land Use Regression Model: **estimate pollution concentration at a given area based on the surrounding land use and traffic characteristics.**	• Allows adaptation to local areas without additional monitoring or data acquisition.	• Depend on density of observations	Briggs *et al.*, 1997; Lebret *et al.*, 2000; Brauer *et al.*, 2002;
Human Inhalation Model: **quantify chemical inhalation from contact with the relevant air pollutants.**	• Can estimate exposure levels for particular receptors and individuals accurately.	• Can be used in areas in with available time activity measurement databases and models.	Burke *et al.*, 2001; Kousa *et al.*, 2002;
Hybrid Exposure Assessment Methods			
Hybrid Models **combine different exposure assessment methods**	• Use existing methods • May combine the strengths of different models or monitoring data	• The integration of different exposure models is difficult to perform • Integration of different scales of measurements	Hoek *et al.*, 2001; Kousa *et al.*, 2002; Isakov *et al.*, 2007

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Appendix A

Supplementary details

This Appendix contains supplementary detail on the methodologies used for the case studies.

London Study:

AEA's Pollution Climate Mapping (PCM) models are used to generate annual mean outputs principally for comparison against legislative values to demonstrate compliance with EU Directives and the UK Government's Air Quality Strategy. In order to represent changing exposure throughout the day the air quality component must be tailored from the standard PCM model annual mean output to 24 separate 1x1km outputs, each representing a typical hour of the day. This diurnal variation can then be used with different population grids to assess the exposure at a city-wide population level at different times of the day.

The model was run using hourly sequential met data for 2007 from the monitoring site at Waddington. This is the same met data that has driven the UK PCM models for a number of years. The dispersion kernels have been generated using ADMS 4.0, a commonly used dispersion model. These dispersion kernels represent a pollution footprint over a set area (33 x 33km) from a source in the central cell.

The influence of different terrain types on dispersion at the land surface results in the need for several types of dispersion kernels with different minimum Monin-Obukhov lengths (the controlling parameter in the dispersion model) dependent on the fabric of the land surface. For example, open grass plains affect dispersion in a very different way than dense urban areas with high-rise buildings. Therefore the model treats different terrain types with the respective dispersion kernel based on an area type map for the London (Figure A-1). Ten area types are considered ranging from highly urbanised at the lower end of the scale, to highly rural at the higher end of the scale. As the map shows, the area types that dominate London within the bounds of the M25 ring road are AT1, 2 and 3. Separate dispersion kernels are made up to represent:

- AT1, 2 and 4
- AT3, 5, 6, 7 and 8
- AT9 and 10

The dispersion kernels are further separated to represent road traffic sources (dispersed low to the ground as a volume source with a height of 10 metres) and non-road traffic sources (dispersed as a volume source with a height of 30m). The result is that 144 separate dispersion kernels are applied within the model to account for road traffic and non-road traffic sources for each of 24 different hours and to account for the 10 different area types.

A diurnal profile of emissions was used for local road traffic sources. A constant profile of emissions has been assumed for non-road traffic local emissions. Thus the contribution from these sources varies during the day as a result of changes in dispersion conditions while the contribution from road traffic sources varies during the day as a result of both the diurnal pattern of emissions and dispersion conditions.

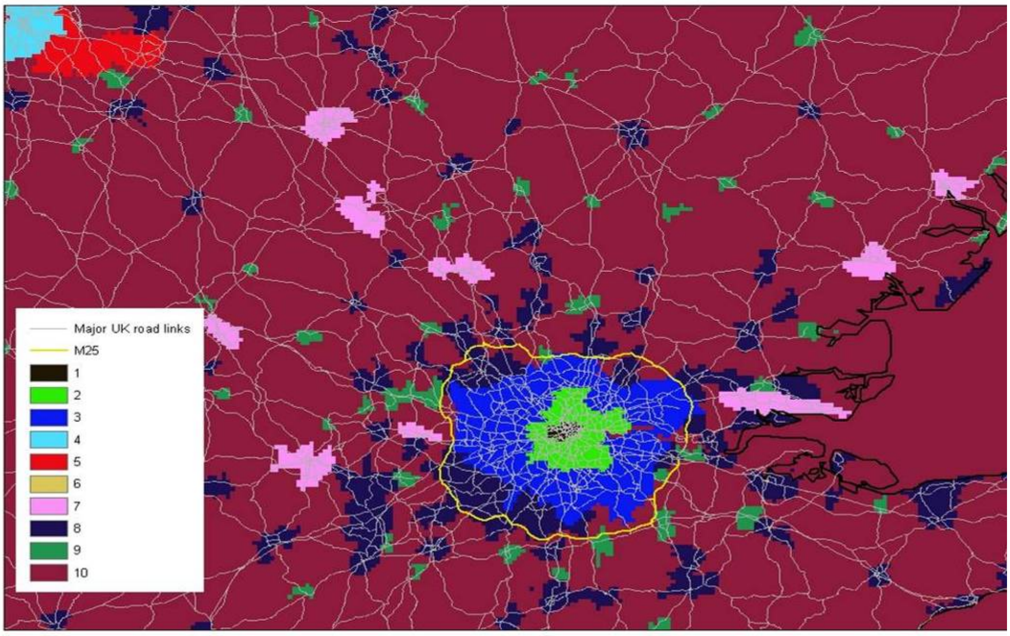


Figure A-1 Area types comprising London and its surrounds

The diurnal variation of other PM sources have not been characterised within the PCM model. These therefore are not dispersed using the time varying kernel approach and assumed to remain constant throughout the day. These include:

- Sea salt
- Secondary inorganic aerosol (nitrates, sulphates and ammonium)
- Secondary organic aerosol
- Point sources
- Long-range transport primary particles
- Iron and calcium rich dusts

A simplified arrangement of 2 PM_{10} maps – to match corresponding population grids representing the hours spent at home (17.00 -08.00) and one to represent the hours spent at work (08.00-17.00) was considered as a possible approach. However, despite the additional complexity and processing time, a separate map was created for each hour to provide more flexibility with regard to scenario analysis in the future because it does not limit the analysis to a range of specified hours but allows the selection of user-specified individual hours to represent time at the working and home locations. The resulting hourly variation maps are shown in the body of the report, in section 4.4.1.

Population patterns

Detailed maps of the contrasting **resident** and **daytime populations** were constructed based on the UK 2001National Census datasets at 1x1 km and at UK Census 'Middle Layer Super Output Area (MLSOA)' spatial resolution. The maps are given below as supporting information:

Combining these distributions with the hourly air quality maps produces exposure maps, summary statistics for which have been given in the body of the report. The associated maps aggregated across the day for both a commuting and a static population are also given here for reference.

Figure A-2: Resident population at MLSOA level in Greater London

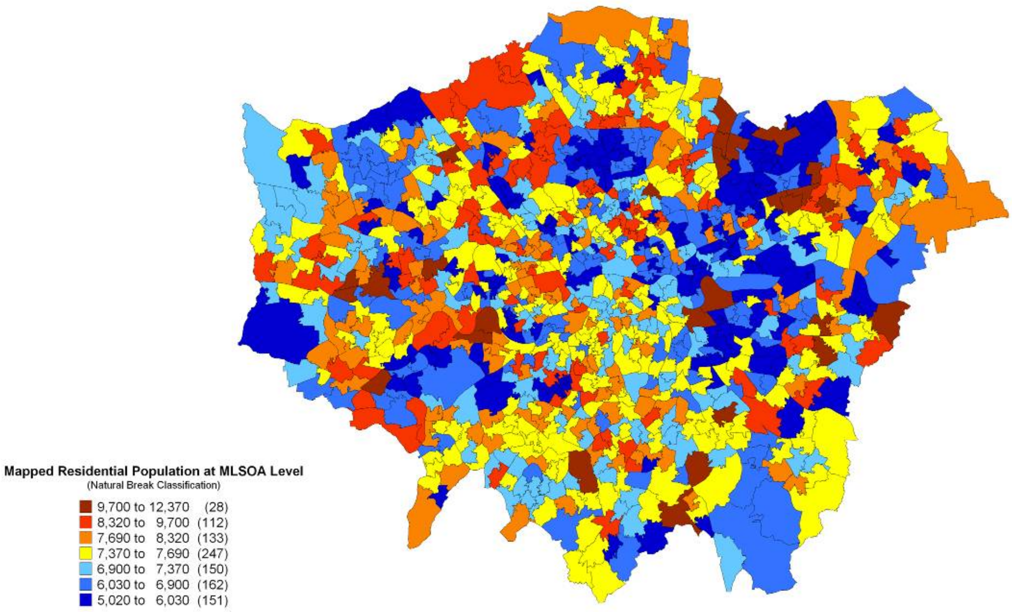


Figure A-3: Daytime population at MLSOA level in Greater London

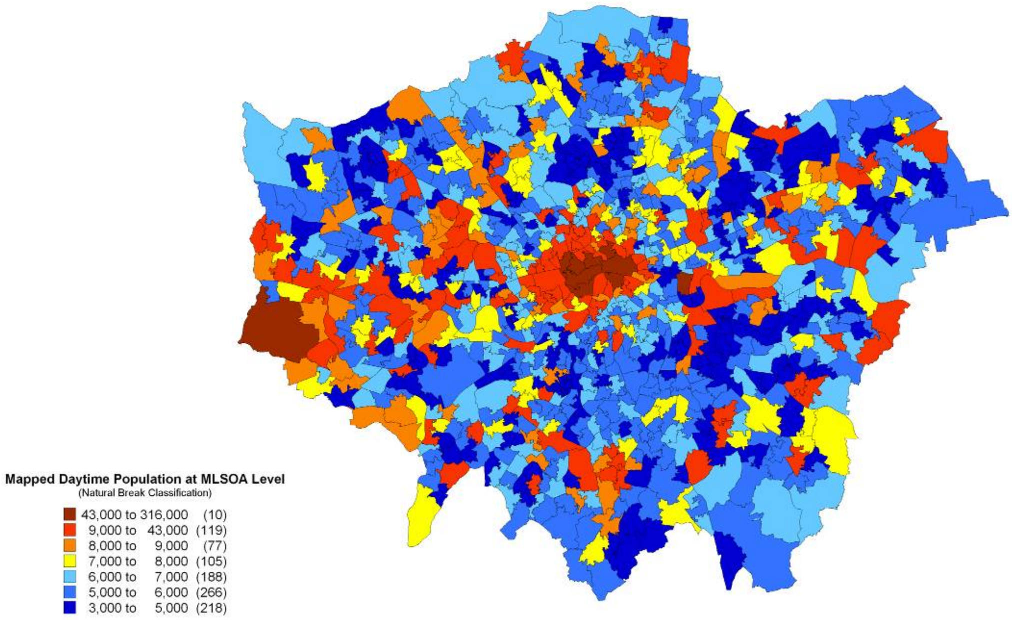


Figure A-4: Residential population at 1km x 1km level in Greater London

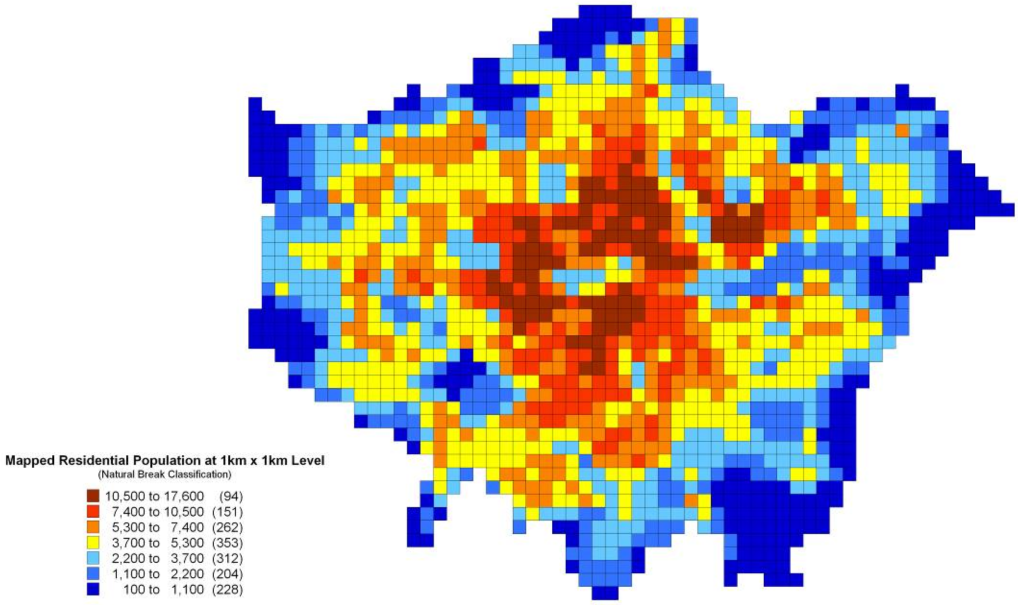
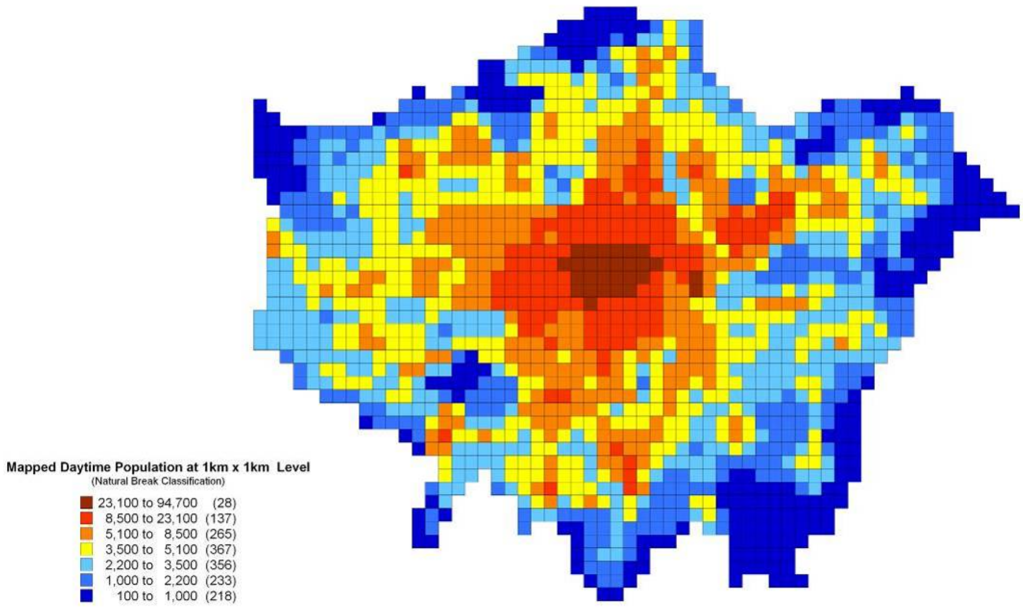
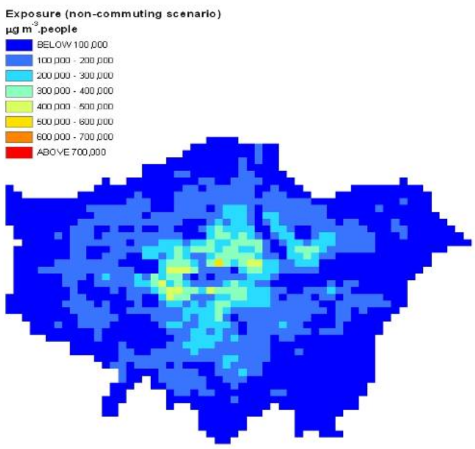
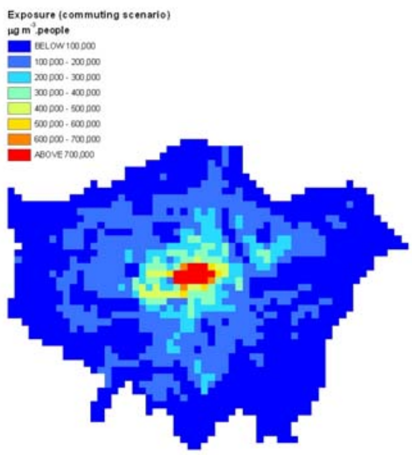


Figure A-5: Daytime population at 1km x 1km level in Greater London



Figures A-6 and 7: Comparison of exposure without (A-6; right) and with (A-7; left) commuting





For analysis and presentation the Greater London area was then subdivided into three – central, inner and outer London. The distribution of metropolitan districts (boroughs) between these subdivisions is:

Central London roughly corresponds to the boundary of the old Central London Congestion Charging Zone (CCZ, launched in February 2003) and is made up of some of the 1km^2 grids in the London boroughs of Camden, City of London, Islington, Lambeth, Southwark, Tower Hamlets, and Westminster.

Inner London includes grid cells from the London boroughs of Camden, Greenwich, Hackney, Hammersmith and Fulham, Haringey, Barnet, Islington, Kensington and Chelsea, Lambeth, Lewisham, Newham, Southwark, Tower Hamlets, Wandsworth, Westminster, Waltham Forest and Redbridge.

Outer London includes grid cells from of some the London boroughs of Enfield, Barking and Dagenham, Barnet, Bexley, Brent, Bromley, Croydon, Ealing, Harrow, Havering, Hillingdon, Hounslow, Kingston-upon-Thames, Merton, Richmond-upon-Thames, Redbridge, Sutton and Waltham Forest.

The description of how population data was handles is given in the main body of the text, in section 4.2.

Moravia-Silesia

The population map for Moravian-Silesia (Figure 2) is based on the data collected by Czech Statistical office from the population registry for January 1st, 2009. The statistical unit of the base data is LAU2. All the maps were constructed in the 1x1 km grid so for the mapping purpose the population data were converted from the LAU2 to the 1x1 km grid using the amounts of area with built-up land in the grid cells. The cells with no built-up land meaning no people live here are filled with white colour in the map. Coloured grid-cells represent inhabited areas. This population map could be understood as the ‘night-time’ population map.

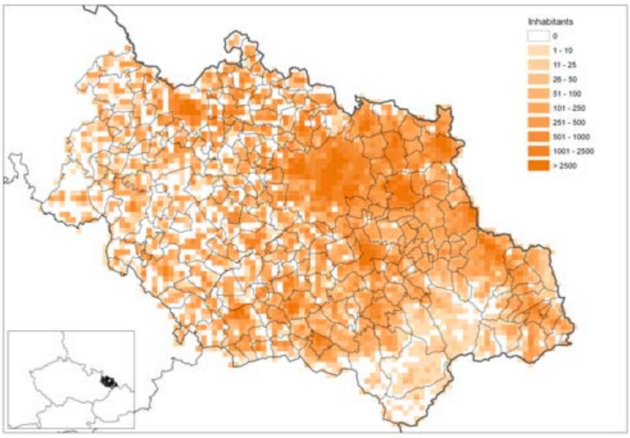


Figure A-8 Map of population density in Moravian-Silesia region

A series of maps of PM10 average concentrations for night, morning, day and evening periods for the second and fourth quarter of year are depictedbelow. In the months January-March and November-December as the local time CET is considered (UTC+1), while in the months April-October it is EET (or summer CET, i.e. UTC+2). The maps were constructed using linear regression of the measured and supplementary data and subsequent spatial interpolation of the regression's residuals (Horálek *et al.*, 2007). As the supplementary data the official annual average PM_{10} map for Czech Republic was used, resp. the mean of two such maps for 2007 and 2008 (CHMI, 2008 and 2009). It was not necessary to distinguish the rural and urban/suburban type of the station: The rural and urban areas are distinguished just in the annual average PM10 map (used as the supplementary data). The linear relation of the measured and supplementary data is quite close (R^2 lies – for the individual time periods - in the interval between 0.73 and 0.98). The interpolation of the regression's residuals was done using IDW (i.e. inverse distance weighted) method. The reason for its use is that this method respects the measured data in the station's points. The uncertainty of the maps (expressed by RMSE from cross-validation) lies (for individual maps) between 1.9 and 4.2 $\mu g.m^{-3}$.

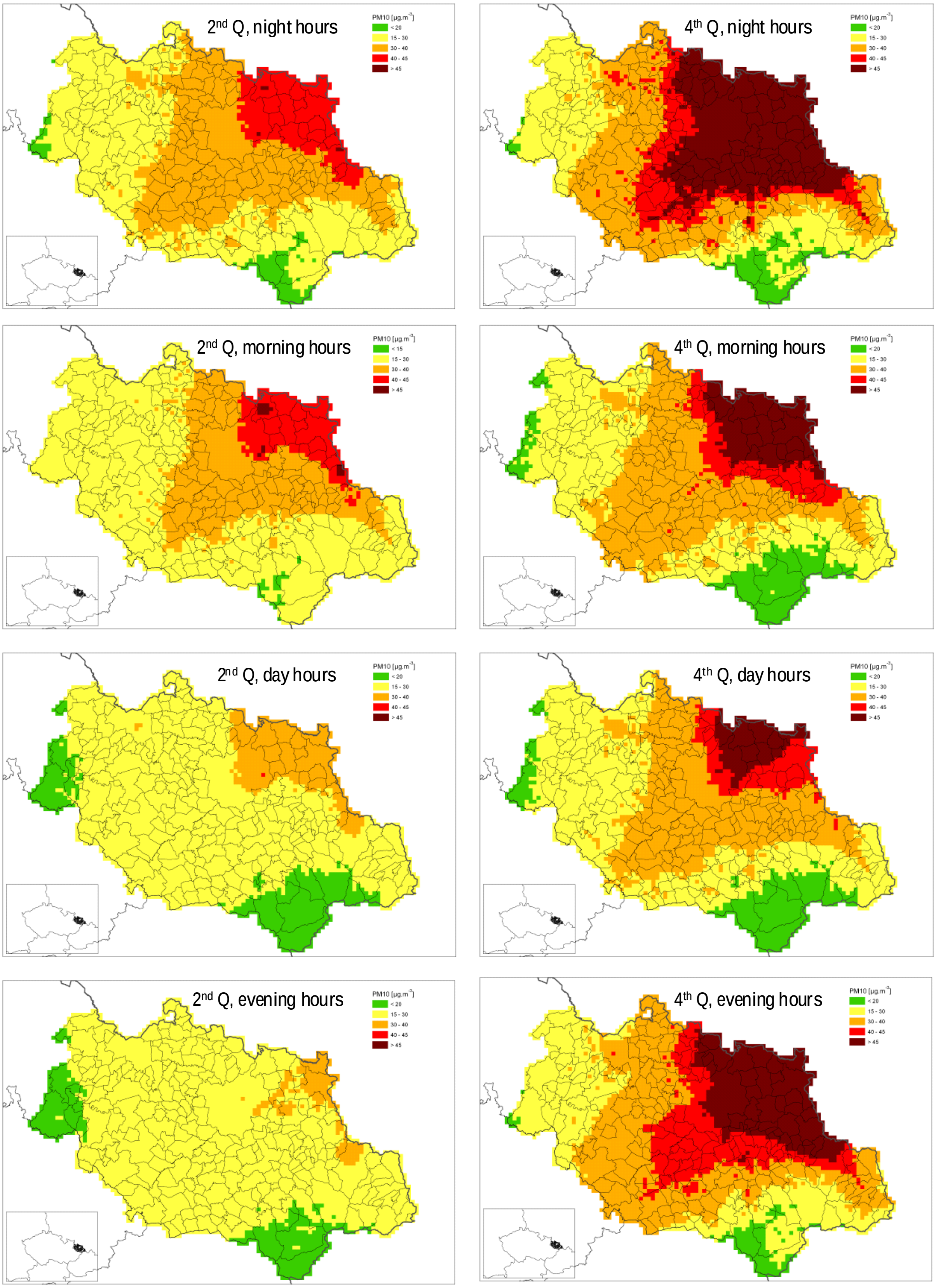


Figure A-9   PM10 average concentrations for night, morning, day and evening periods in the second and fourth quarters

<u>Athens study</u>

National Census data for 2001 from the National Statistical Service of Greece (NSSG) were employed revealing a residential population of around 4 million. According to the 2001 census data, 35% of the GAA is covered by residential buildings, 21% of area is by roads (in contrast to the municipality of Athens where the percentage is around 26%), 7% by industry and only 5% for recreation activities. The data employed for the population movement within GAA were also provided by the NSSG. Figure A10 shows the locations of municipalities with populations of over 1500 inhabitants, and Figure A11 shows the estimated population variation as a percentage of the residential population on the basis of the 2001 Census, for those municipalities. It should be noted that these municipalities of the GAA, are located primarily in the Athens – Piraeus area and host the majority of the population .

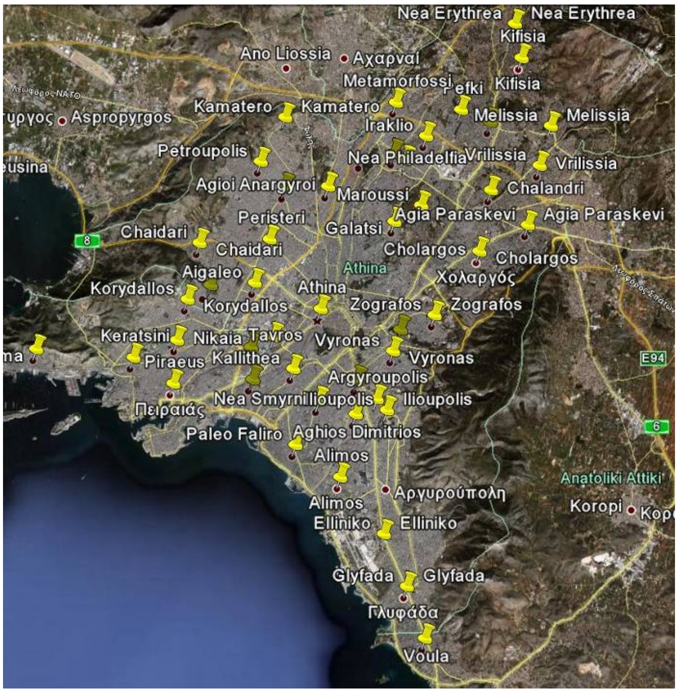


Figure A10: Locations of municipalities with population over 15000 inhabitants in GAA

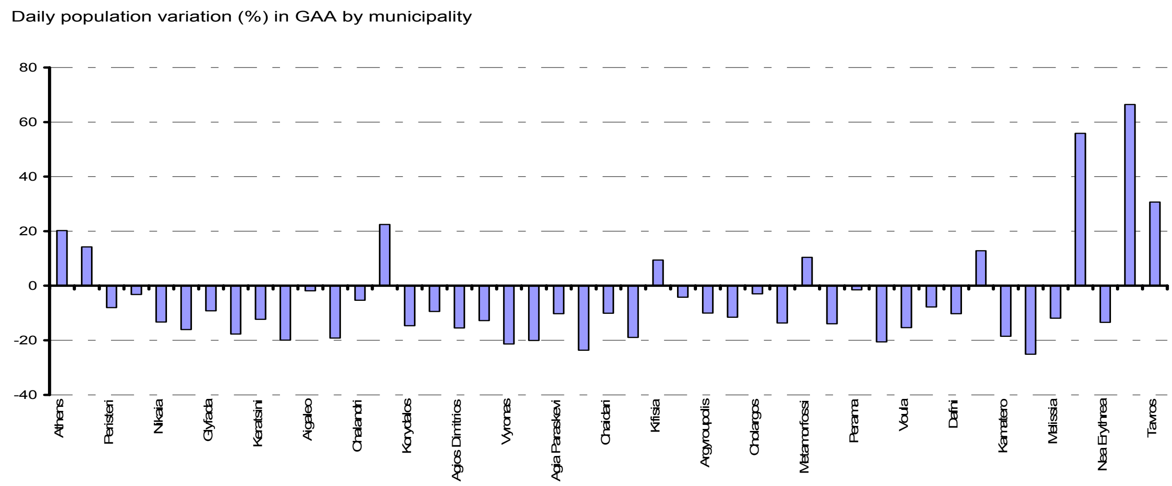


Figure A11: Daily population change in the Attica region for municipalities with population over 15000 inhabitants

The OFIS model

The OFIS model belongs to the European Zooming Model (EZM) system, a comprehensive model system for simulations of wind flow and pollutant transport and transformation (Moussiopoulos, 1995) and was developed to serve a twofold aim; (i) allowing authorities to assess urban air quality by means of a fast, simple and still reliable model and (ii) refining a regional model simulation by estimating the urban sub grid effect on pollution levels.

OFIS simulates concentration changes due to the advection of species and chemical reactions in each cell of the computational domain. The first vertical layer extends up to 90m (in accordance with the EMEP model, which provides the boundary conditions), while the second one extends up to the mixing height, thus varying with time. Emission data are inserted into the model in the form of gridded emission inventories. Emissions are calculated for each OFIS cell by properly taking into account the emission density of the underlying fine-scale inventory. Biogenic emissions are also taken into account for rural areas. Due to the modular structure of OFIS, chemical transformations can be treated by any suitable chemical reaction mechanism, the default being the EMEP MSC-W chemistry (Arvanitis and Moussiopoulos, 2003).

Inflow boundary conditions can be derived from available monitoring data or, preferably, taken from results of a regional scale model. Meteorological input may be derived and fed into the model from either available measurements or the output of a larger scale model. The numerical solution of the equation system is based on a variable step, second order BDF formula and a Gauss-Seidel iterative technique (Verwer, 1994). The OFIS model can take advantage of the frequency by which boundary conditions and meteorological data from larger scale models usually become available. An appropriate parameterisation for wet removal (Scott, 1979), an important physical process especially with regard to particulate matter, is also part of the model. Parameterisation of the wet deposition processes includes both in-cloud and sub-cloud scavenging of particles, including $PM_{2.5}$ and PM_{10}. While sub-cloud scavenging is taken into account in both layers, in-cloud scavenging is only applied on the model's second layer. The model simulates separately each day of, typically, one year.

Although the most densely populated part of the GAA extends over an area of 50 km × 50 km, for the needs of the GAA the modeled domain extended over an area of 150 km × 150 km in order to account for potential effect of suburban areas, which are not usually considered as part of the GAA. OFIS was run for the reference year 2000, using as inflow boundary conditions for the meteorology input from the EMEP model and calculated gridded emissions for all SNAP sectors. As no modeling work for the needs of this year's work was foreseen, the analysis in the current report as regards the assessment of the air pollution over GAA is based in last year's numerical results. Further information can be found at Barret *et al.*, 2008 and Moussiopoulos *et al.*, 2007.

Air quality and exposure maps

Analysis of the population exposure is based on numerically assessed concentration levels for PM_{10} over the cells of the domain approximating the GAA. Results for the average daily variation of the PM_{10} concentration levels were extracted both over the entire domain of 150 km × 150 km (for practical purposes it will be referred to as "large"), as well as the main urbanized part of Attica which corresponds to the aforementioned area of 50 km × 50 km (for practical it will be referred to as "small"). A comparison between the estimated daily variation over the large and the small domains is shown in Figure A12. The comparison reveals that the average daily variation concentration levels estimated over the large domain are around 5% lower compared to the average daily variation of the concentration levels estimated over the small domain. The reason for that is that the main contributor to air pollution is Athens and the surrounding areas which lie at close proximity. The average daily variation of PM_{10} concentrations was estimated for each cell of the domain under consideration at 1 km × 1 km resolution and a total of 24 air quality maps were created, each one corresponding to an hour of the day. In Figure A13, yearly averaged PM_{10} concentration level maps over the domain under consideration are presented for selected hours of the day. The same scaling was applied to all maps.

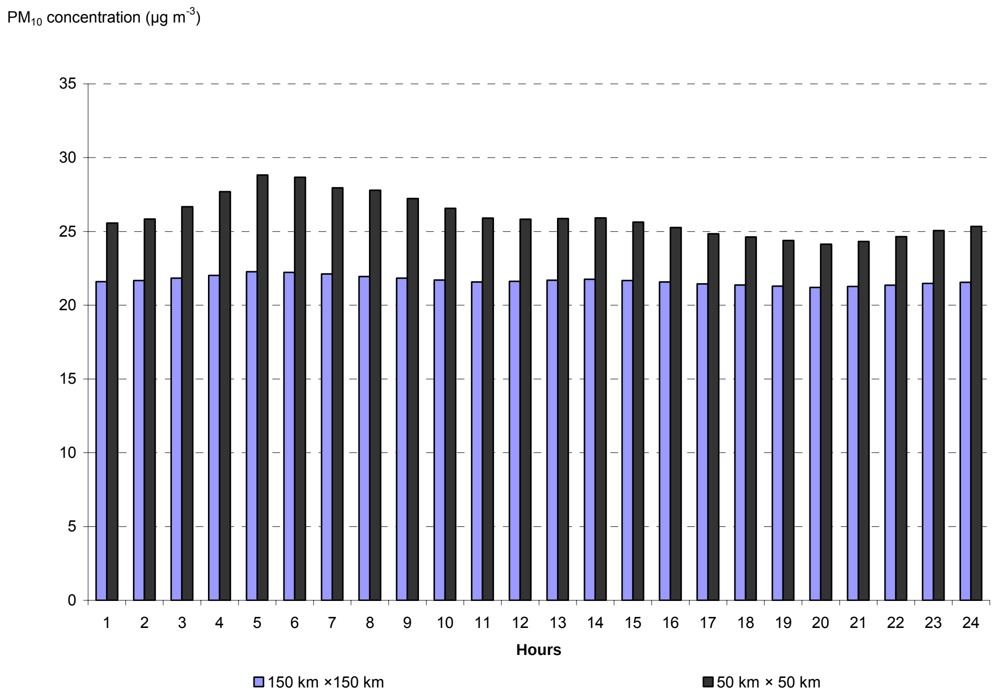
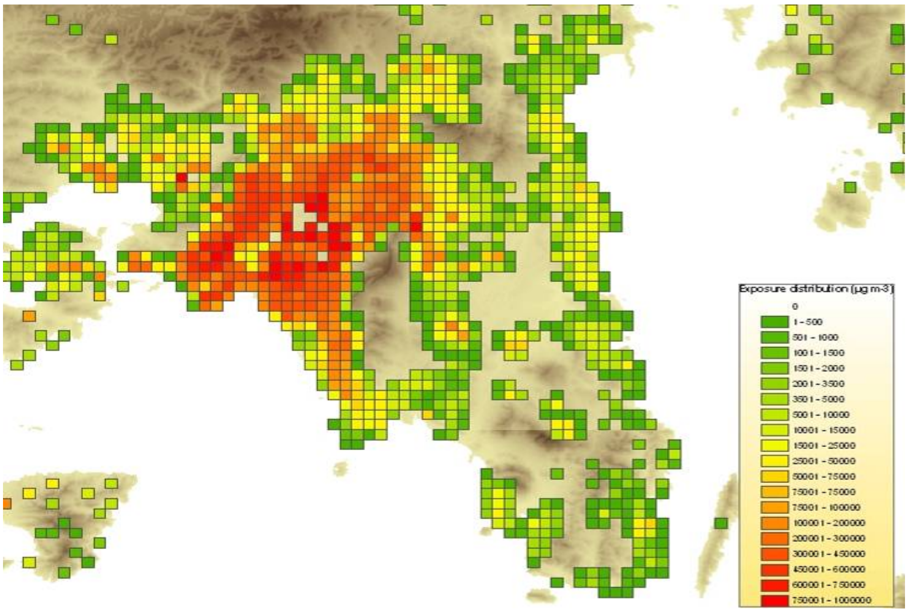


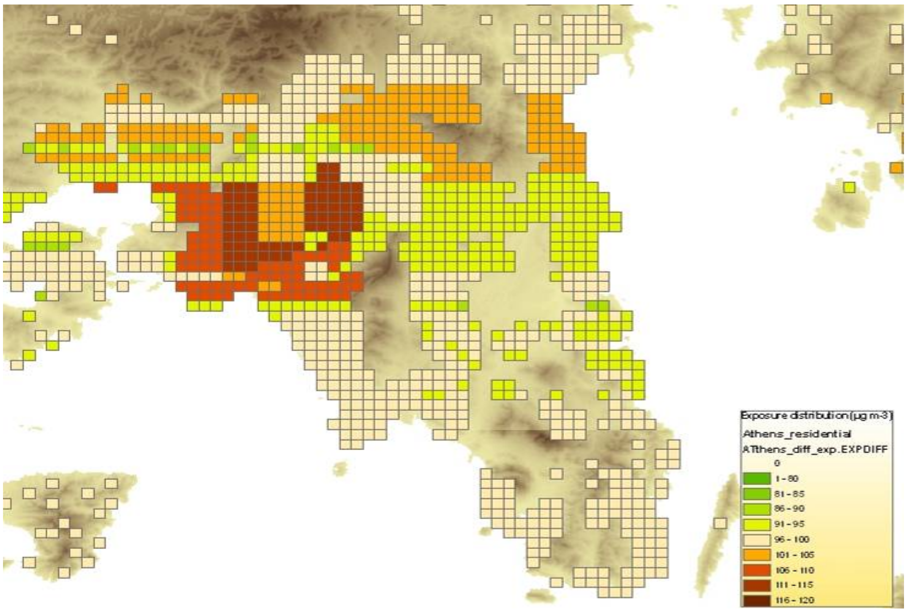
Figure A-12: Daily variation of the average PM10 concentration level over the domain under consideration

Using GIS spatial analysis, these maps were then combined with population flow data for the different municipalities comprising the GAA described in the previous section, in order to calculate the population exposure by taking into account intra urban commuting. The estimated effects of the commuting of population are presented in the form of difference maps between the annual average exposure and the annual average hourly exposure for each hour of the day.

Figure A13 illustrates the annual average exposure based on the residential population distribution, while Figure A14 shows the estimated daily spatial variation upon this residential exposure arising as a consequence of commuting, expressed as a percentage of the calculated population exposure for the static population (non-commuting) scenario.



FigureA.13: Annual average population exposure distribution for the non – commuting scenario



FigureA14: Daily annual average population exposure variation (%) due to commuting for work